Winter Measurements of Heavy-duty Vehicles to Characterize the Cold Temperature Effectiveness of Selective Catalytic Reductions Catalyst in Controlling Oxide of Nitrogen Emissions

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September 2021

Executive Summary

The University of Denver conducted a winter emissions measurement campaign for heavy-duty vehicles (HDV) at the southbound Perry Port of Entry along I-15 ~5 miles south of Brigham City Utah. The remote sensor used in this study measures the ratios of CO, HC, NO, NH_3 and NO_2 to CO_2 in the exhaust of the passing trucks. From these ratios, we can calculate the fuel specific (grams of pollutant/kg of fuel consumed) emission factors for CO, HC, NO, NH_3, NO_2 and NO_x ($\equiv$ NO + NO_2) in the exhaust of the passing vehicles. The system used in this study was configured to determine the speed and acceleration of the vehicle, and was accompanied by a video system to record the license plate of the vehicle and, from this record, the vehicle's registration information. In addition exhaust thermographs were taken with an infrared camera (Thermovision A20, FLIR Systems) for estimating the exhaust temperatures of the trucks with elevated exhaust pipes leaving the weigh station.

In the Salt Lake City region fine particulates accumulate during periods of low winds and thermal inversions where warm air aloft traps the air mass against the ground and mountains to the west. This is especially problematic during the winter months when thermal inversions can last over many days allowing particulates to accumulate. As a consequence, the region has a serious designation for violation of the 24-hour $PM_{2.5}$ standard from increases in ammonium nitrate aerosols. These particles are formed in the atmosphere from local and transported sources of NH_3 and nitrate with the main source of nitrate expected to be from NO_x emissions. HDV are a significant source of NO_x emissions in the Salt Lake region and their wintertime emissions performance has not been investigated as the majority of HDV emission studies over the last two decades have been performed during warm weather months in California.

Over parts of six days (Sunday Dec. 6 to Friday Dec. 11, 2020) the University of Denver collected exhaust emission measurements with the FEAT remote sensor operated in an elevated configuration for ~22 hours to capture exhaust plumes from HDV with elevated exhaust stacks and at ground level for ~10 hours to sample trucks with ground level pipes. Most measurements were collected during daylight hours (~28 hrs.), however, on two evenings some measurements were collected after dark. The campaign resulted in the successful measurement of 1694 vehicles, the majority of which were class 7 & 8 heavy-duty vehicles (1591) and the remaining 103 measurements from medium-duty vehicles. Vehicles from 37 different states and Canada were sampled with the largest numbers from Utah (35.5%) and Idaho (13.9%). A database was compiled containing 1694 records with the emission measurements, vehicle registration information and additional vehicle specific information obtained from decoding the Vehicle Identification Number. The database, as well as others compiled by the University of Denver, can be found at https://digitalcommons.du.edu/feat/.

For the 1591 class 7 & 8 heavy-duty vehicles the mean CO, HC, NO, NH_3, NO_2 and NO_x emissions were 5.8 ± 1.5 gCO/kg of fuel, -0.08 ± 0.07 gHC/kg of fuel, 11.5 ± 1.3 gNO/kg of fuel, 0.08 ± 0.06 gNH_3/kg of fuel, 0.67 ± 0.09 gNO_2/kg of fuel, 18.5 ± 2.0 gNO_x/kg of fuel and 0.6 ± 0.1 %IR Opacity respectively. The average chassis model year was 2014.2 and the Utah plated vehicles were 2.4 model years older than the out of state fleet (2012.8 versus 2015.2). Outside air temperatures were recorded at the site at 5 minute intervals during all 6 days of sampling with ambient temperatures ranging from -7 to 10°C with an average of 3.8°C during the HDV measurements.

Fuel specific NO_x emissions were found to be significantly higher than the most recent warm weather measurements collected at a weigh station in California in 2017. Figure ES1 graphs the fuel specific NO_x emissions by chassis model year comparing the 2020 Utah and 2017 California HDV measurements. The uncertainties are standard error of the mean determined from the daily measurements. The differences in this comparison are obvious as the California measurements show a significantly slower increase in NO_x emissions between the newest and oldest HDV. In addition NO_x emissions observed in Utah are also higher at both ends of the age distribution as well.

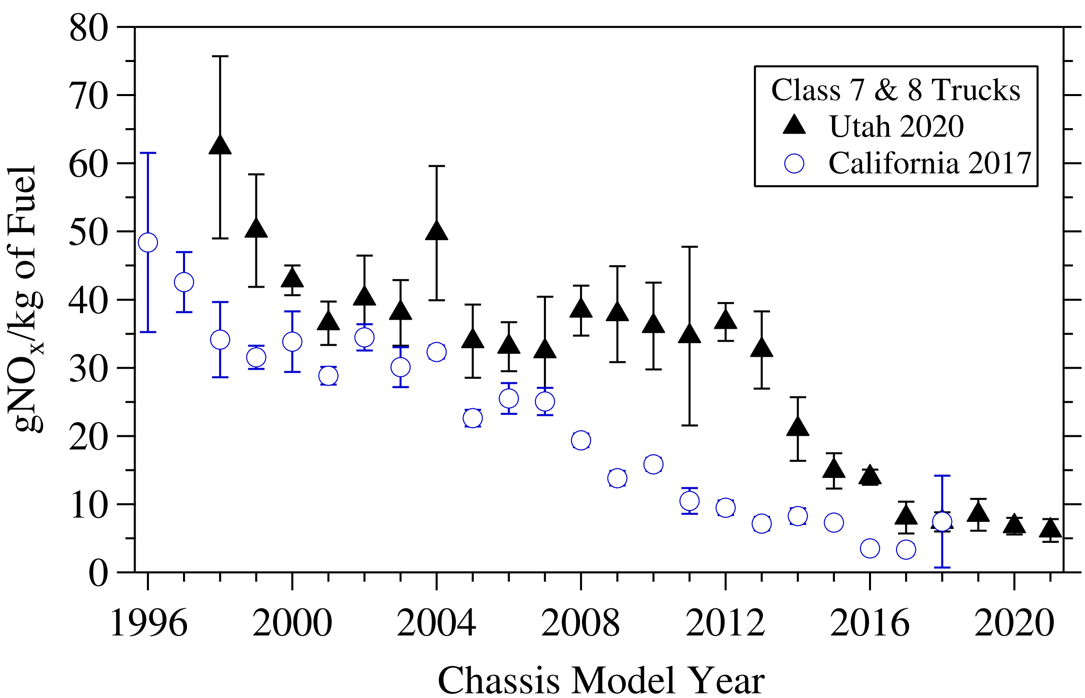

Figure ES1. HDV gNO_x/kg of fuel versus chassis model year for the 2020 Utah measurements (triangles) and the 2017 California measurements (circles). Uncertainties are standard error of the mean calculated from daily means.

2017 and newer model year HDV form have the lowest NO_x emissions that within the measurement uncertainties are all similar indicating little to no emissions deterioration on average over the five years. The NO_x emission then increase between the 2016 and 2013 model year vehicles to a second and significantly higher NO_x emission levels where again within the measurement uncertainties there are no real differences from the 2013 to the 2000 chassis model year HDV. Under winter conditions at this site we find that chassis model years 2011 to 2013 HDV have on average completely lost any previous benefit gained from their NO_x after-treatment systems.

While it's difficult to unequivocally ascribe the increased NO_x emissions to a specific cause with only a single data set, analysis of the oldest and newest trucks in the fleets suggests a temperature effect that increases NO_x emissions between approximately less than 10% and 25% for the oldest and newest vehicles. Using the IR thermographs from only the elevated exhaust pipes we found that the estimated mean pipe temperature of 92° in Utah were 18°C colder than similar readings from the California study (110°C) again suggesting colder temperatures for the exhaust after-treatment systems. We did find that the lower temperatures did not result in an increase in the number of SCR systems that appeared to be completely inactive but we believe that the reduced temperatures likely lowers the NO_x conversion efficiencies thus increasing NO_x emissions in trucks found in the middle percentiles. Increases in the NO_x emissions of the models with the oldest SCR systems (chassis model years 2011 - 2013) appear to most likely be caused by significant emissions deterioration in their after-treatment systems.

Comparison of the HDV NO_x emission measurements with the newest U.S. Environmental Protection Agencies MOVES3 and California's EMFAC2021 models found significant differences in 2004 and newer chassis model year vehicles. Figure ES2 compares the fuel specific NO_x emissions between the Utah 2020 measurements, MOVES3 estimates for Box Elder county Utah and an EMFAC2021 Statewide California emissions winter estimate. Uncertainties for the Utah measurements are standard error of the mean determined using the daily means. MOVES3 engine model years have had one year added to them to convert to chassis model year to match the measurements and EMFAC2021 estimates. The Utah measurements have significantly higher NO_x emissions for all model years except for the 2003 and older chassis model years where there is good agreement. The reduction in emissions predicted by the models starting with the 2004 chassis model year vehicles and the subsequent reduction to very low levels after the 2014 chassis model year vehicles does not occur in the Utah measurements.

Using the Utah age distribution and the MOVES3 emission factors by chassis model year resulted in mean NO_x emissions of 18.5 ± 2.0, 10.4 and 7.3 for the Utah measurements, MOVES3 and EMFAC2021 estimates respectively. The MOVES3 model was run for Utah in December of 2020 but note that MOVES3 does not make a distinction between summer and winter. The factor of 1.8 under prediction by the model likely means that the winter NO_x inventory for the Salt Lake region is also under estimated.

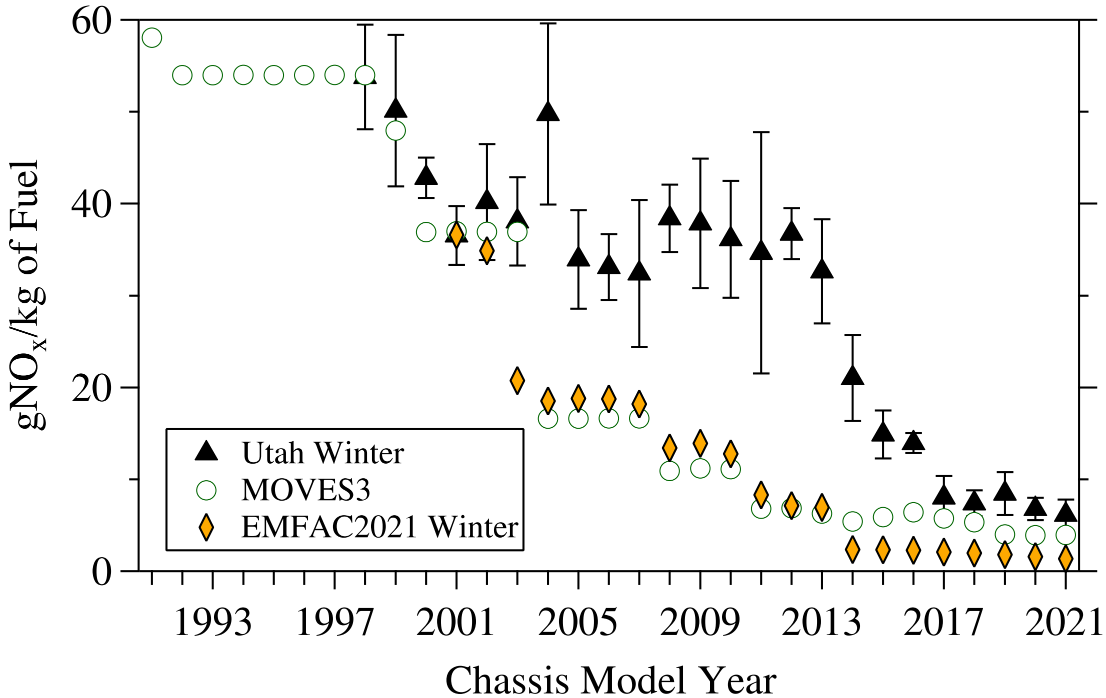

Figure ES2. Fuel specific NO_x emissions versus chassis model year for the Utah measurements, the MOVES3 estimates for Box Elder County and EMFAC2021 winter estimates for California. Uncertainties are standard error of the mean calculated using the daily means.

Because of the significant differences between the California and the U.S. heavy-duty fleet we do not expect the EMFAC2021 model estimates to be necessarily representative of the Utah fleet. It predicted a gNO_x/kg of fuel of 7.3 gNO_x/kg of fuel and an emission versus chassis model year relationship that is reasonably similar to the predictions by the MOVES3 model with lower emissions for the newest chassis model year vehicles due to the MOVES3 model including a significant number of "Glider" vehicles. Comparison of fuel specific NO_x emissions estimated by EMFAC2021 in the winter or summer does result in mean NO_x emissions being 34% higher during the winter scenario.

We also were able to show that some of the observed NO_x emission deterioration observed in the 2011 - 2013 chassis model year vehicles is likely related to the voluntary recall of Cummins engines and after-treatment systems for defective SCR systems. In addition there is a small population of HDV identified as "Gliders" (23/1591) which have significantly higher NO_x and %IR Opacity than similar chassis model year vehicles.

Introduction

In the United States in 2019 it is estimated that trucking moved more than 70% of the country's freight.[1] The overwhelming majority of the trucks moving this freight are diesel powered. Nitrogen oxides (NO_x = NO + NO_2) and particulate matter (PM) are major constituents of diesel exhaust from heavy-duty vehicles (HDV) and can lead to serious air quality and damaging health effects.[2-4] As a result since the middle 2000's regulators have focused heavily on reducing both of these pollutants from the exhaust of HDV. Both Federal and California regulations have resulted in the certification standards for both NO_x and PM being lowered by an order of magnitude. First for PM beginning in 2007 engines (2008 chassis model year typically) with the addition of diesel particulate filters (DPF), which are a ceramic size exclusion filter that prevents soot particles above a certain size from exiting the exhaust system.[5] This was followed with reductions in the NO_x standards for 2010 engines (2011 chassis model year) resulting in the addition of selective catalytic reduction (SCR) after-treatment systems. These systems reduce NO_x emissions by reacting it with ammonia (NH_3), created from thermalizing a urea and water solution, to form nitrogen.[6]

HDV in the U.S. serve many purposes but typically are high mileage vehicles whose diesel engines last for long periods of time. With this in mind the State of California has instituted a number of regulations that force the early retirement of older technology HDV with the aim of significantly reducing PM and NO_x emissions from this segment of the fleet.[7, 8] Other states have not followed suit instead relying on natural attribution and fleet turnover to modernize the fleet. As a result of the considerable effort and expense that California has committed a number of data collection campaigns have been funded to research their progress and there have been few if any measurements of HDV emissions outside California since 2005.[9-12] Despite significant progress in reducing both PM and NO_x emissions HDV fuel usage continues to grow and they are still a major source of both pollutants with a larger influence now found outside of California.[13]

In the Salt Lake City region fine particulates accumulate during periods of low winds and thermal inversions where warm air aloft traps the air mass against the ground and mountains to the west. This is especially problematic during the winter months when these periods can last over many days allowing particulates to accumulate. As a consequence, the region has a serious designation for violation of the 24-hour $PM_{2.5}$ standard. In 2017, the Utah Winter Fine Particulate Study (UWFPS) was conducted to investigate the sources, composition, and chemistry of the fine particulates.[14] Figure 1 shows the composition of submicron particles ($PM_{1.0}$) as measured by the National Oceanic & Atmospheric Administration's (NOAA) Twin Otter during the UWFPS, which was instrumented with an aerosol mass spectrometer. As illustrated, during cold-pool air pollution episodes, the dominant fraction of $PM_{1.0}$ is nitrate (blue pie ~ half of total). Nitrate (NO_3^-) reacts with ammonia (NH_4^+) to form ammonium nitrate aerosol. When taken together, ammonium nitrate (blue + orange pies) accounts for ~80% of the $PM_{1.0}$ during episodes.

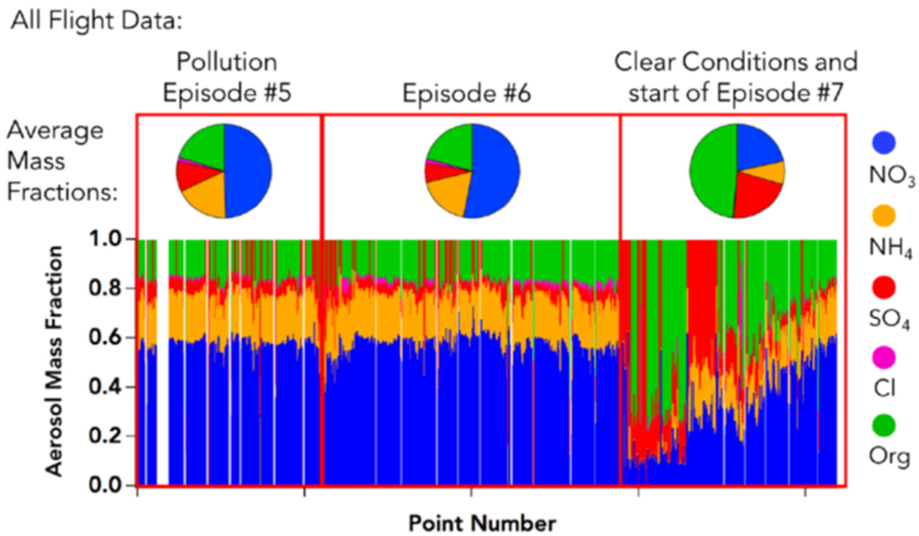

Figure 1. Composition of submicron particulate matter ($PM_{1.0}$) in Salt Lake City during the 2017 Utah Winter Fine Particulate Study (Brown et al., 2017). During cold-pool episodes, nitrate is the dominant fraction of the $PM_{1.0}$ mass.

Ammonium nitrate aerosols are formed in the atmosphere from local (e.g., vehicles) and transported (e.g., agriculture) sources of NH_3 and nitrate with the main source of nitrate expected to be from NO_x emissions. In addition the performance of diesel engines at colder temperatures has become important as recent research has shown that peak wintertime $PM_{2.5}$ concentration maxima have not been reduced along with the summertime maxima[15]. In addition, recent research from Europe has shown a NO_x temperature dependence for light-duty diesel vehicles with increasing emissions at lower temperatures, even for vehicles with the newest after-treatment systems.[16] Current computer models only include temperature dependencies for air conditioning operation at high temperatures and likely underestimate wintertime NO_x levels. Furthermore, preliminary modeling by the NOAA Chemical Sciences Laboratory over the Continental US illustrates that current inventories of US NO_x emissions perform well for summertime (Figure 2, left: model bias = -3 to -17%) and under-predict significantly in the wintertime (Figure 2, right: model bias = -32 to -59%). The model was simulated with NO_x emissions updated to 2018 utilizing a Fuel-based Inventory of Vehicle Emissions (FIVE) developed by McDonald et al. for mobile sources, continuous emissions monitoring system data for power plants, and the National Emissions Inventory 2014 for all other sources[17]. The model under-predictions are even more pronounced over the urban areas, such as New York City, suggesting a missing or under-accounted urban source.

The majority of HDV emission studies performed over the last two decades have not been performed during the cold winter months. A tunnel and two near-road measurement campaigns have been performed recently during the winter months all showing higher NO_x emissions during the winter season.[18-20] These studies strongly suggest a seasonal dependence in NO_x

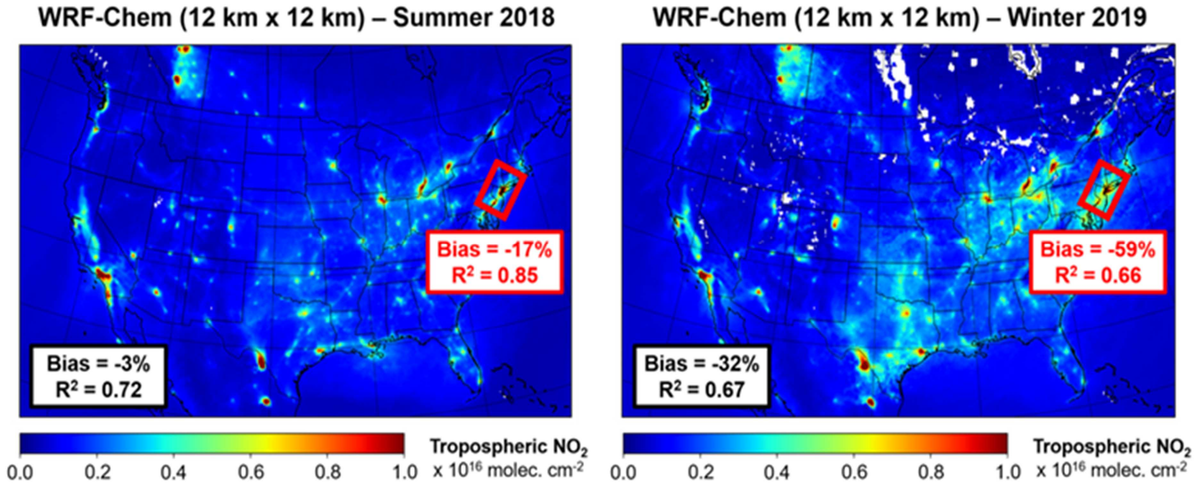

Figure 2. Modeling Continental US NO_x emissions in the Weather Research and Forecasting with Chemistry (WRF-Chem) Model and evaluations with tropospheric NO_2 satellite columns from TROPOMI for (left) summer and (right) winter.

emissions but lack the detailed heavy-duty vehicle fleet characterization necessary to fully explain the differences observed.

In Salt Lake County it is estimated that on-road mobile sources account for approximately 45% of the NO_x emissions emitted each year and that diesel vehicles are responsible for approximately 45% of the total on-road mobile sources, despite the HDV fleet being less than 5% of the vehicles. With this sector representing a significant contributor to the valley's NO_x emissions inventory and the importance of these emissions in the formation of ammonium nitrate aerosols, a critical research question is to find out what if any differences exist in HDV emissions during winter conditions. This project aims to make some of the first winter time emission measurements from HDV in the U.S. where the age distribution of the fleet will be completely characterized. This is important as the range of emissions technology in the current fleet is large and varied and determining what if any effects winter conditions have on different ages of HDV is as important as documenting the actual changes.

Experimental

The Fuel Efficiency Automobile Test or FEAT is a spectroscopic sensor developed at the University of Denver for remotely measuring the pollutants in motor vehicle exhaust, and has previously been described in the literature.[21-25] The instrument consists of a non-dispersive infrared (IR) component for detecting CO, CO_2, HC, and percent opacity, and two dispersive ultraviolet (UV) spectrometers for measuring NO, NO_2, sulfur dioxide (SO_2), and NH_3. The source and detector units are positioned on opposite sides of the road in a bi-static arrangement. Collinear beams of IR and UV light are passed across the roadway into the IR detection unit, and are then focused onto a dichroic beam splitter, which serves to separate the beams into their IR and UV components. The IR light is then passed onto a spinning polygon mirror, which spreads the light across the four infrared detectors: CO, CO_2, HC, and reference (opacity is determined by plotting reference vs. CO_2). The UV light is reflected off the surface of the beam splitter and is focused onto the end of a quartz fiber-optic cable, which transmits the light to dual UV spectrometers. The UV spectrometers are capable of quantifying NO, NO_2, SO_2, and NH_3 by measuring absorbance bands in the regions of 205 - 226 nm, 429 - 446 nm, 200 - 220 nm, and 200 - 215 nm, respectively, in the UV spectrum and comparing them to calibration spectra in the same regions.

The exhaust plume path length and density of the observed plume are highly variable from vehicle to vehicle, and are dependent upon, among other things, the height of the vehicle's exhaust pipe, wind, and turbulence behind the vehicle. For these reasons, the remote sensor directly measures only ratios of CO, HC, NO, NO_2, NH_3, SO_2 to CO_2 (i.e. CO/CO_2, HC/CO_2, NO/CO_2 etc.). Appendix A provides a list of the criteria for valid/invalid data. These measured ratios can be converted directly into grams of pollutant per kilogram of fuel. This conversion is achieved by first converting the pollutant ratio readings to the moles of pollutant per mole of carbon in the exhaust from the following equation:

$$\frac{\text{moles pollutant}}{\text{moles C}} = \frac{\text{pollutant}}{CO + CO_2 + 3HC} = \frac{(\text{pollutant}/CO_2)}{(CO/CO_2) + 1 + 6(HC/CO_2)} = \frac{(Q,2Q',Q'')}{Q+1+2*3Q'}$$

Q represents the CO/CO_2 ratio, Q' represents the HC/CO_2 ratio and Q" represents the NO/CO_2 ratio. Next, moles of pollutant are converted to grams by multiplying by molecular weight (e.g., 28 g/mole for CO), and the moles of carbon in the exhaust are converted to kilograms by multiplying the denominator by 0.014 kg of fuel per mole of carbon in fuel, assuming the fuel is stoichiometrically CH_2. The HC/CO_2 ratio uses a factor of two (Singer factor) times the reported HC because the equation depends upon carbon mass balance and the NDIR HC reading is about half a total carbon FID reading.[26] For natural gas vehicles the appropriate factors for CH_4 are used along with a Singer factor of 3.13. Grams per kg fuel can be converted to g/bhp-hr by multiplying by a factor of 0.15 based on an average assumption of 470 g CO_2/bhp-hr.[27]

Negative fuel-specific emissions can be seen in some of the results presented, which does not mean that the vehicles were cleaner than the background air but reflects true zero-emissions that are reported as negative values, as explained below. FEAT's basic units of measurement are molar emission ratios of pollutants (e.g. CO/CO_2, HC/CO_2, NO/CO_2 etc.) with the ratios being the linear regression slopes of the pollutant versus CO_2 measured 50 times during a half-second or 100 times during a one second measurement. An "ideal" zero emission measurement would have a correlation plot with a slope of zero. In real-world measurements, however, instrument and environmental noises inevitably result in positive slopes in some true zero-emission plumes and negative slopes in other true zero-emission plumes. In fact, properly calibrated instruments are expected to result in a zero-centered normal distribution for all the true zero-emission plumes where half of the measurements are positive and half are negative and they average zero. For this reason, we preserve the negative values in the FEAT database and include those values in this analysis to offset the positive tail of the zero-emission distribution, so that the sample average is not biased toward positive.

The FEAT detectors were calibrated, as external conditions warranted, from certified gas cylinders containing known amounts of the species to be measured. This ensures accurate data by correcting for ambient temperature, instrument drift, etc. with each calibration. Because of the reactivity of NO_2 with NO and SO_2 and NH_3 with CO_2, three separate calibration cylinders are needed: 1) CO, CO_2, propane (HC), NO, N_2 balance; 2) NO_2, CO_2, air balance; 3) NH_3, propane, balance N_2. Since fuel sulfur has been nearly eliminated in US fuels, SO_2 emissions are generally below detection limits. While vehicle SO_2 measurements are routinely collected and archived for each data campaign, since 2012 we have not calibrated these measurements and they are not included in the discussion of these results.

The FEAT remote sensor is accompanied by a video system that records a freeze-frame image of the license plate of each vehicle measured. The emissions information for the vehicle, as well as a time and date stamp, is also recorded on the video image. The images are stored digitally, so that license plate information may be incorporated into the emissions database during post-processing. A device to measure the speed and acceleration of vehicles driving past the remote sensor was also used in this study. The system consists of a pair of infrared emitters and detectors (Banner Industries) which generate a pair of infrared beams passing across the road, six feet apart and approximately five feet above the surface. Vehicle speed is calculated from average of two times collected when the front of the tractors cab blocks the first and the second beam and the rear of the cab unblocks each beam. From these two speeds, and the time difference between the two speed measurements, acceleration is calculated, and reported in mph/s. An additional set of an emitter and detector are used to cue the FEAT detectors measurement of each truck plume. Appendix B defines the database format used for the data set.

Measurements were collected at the Utah Department of Transportation Perry Port of Entry located ~5 miles south of Exit 362 to Brigham City, UT on the southbound lanes of I-15. Measurements were collected from Sunday, December 6, to Friday, December 11, 2020 generally between the hours of 8:00 and 19:00 on the exit lane reentering southbound I-15. Figure 3 shows a satellite photo showing the relative location of the Perry Port of Entry to the Salt Lake City area (A, left image) and a second photo (B, right image) highlighting the layout of the Port of Entry and showing the approximate emissions sampling location. The station has two lanes that trucks exiting the highway are directed into. The inside lane has the scales and a posted speed limit of 3 mph while the outside lane has a posted speed limit of 20 mph. It is 750m from the initial highway exit point to our measurement location with approximately 175m subject to the speed limits. After the lanes merge the trucks are allowed to accelerate to highway speeds for their return to the freeway. At an average speed of 10mph a truck will spend less than 5 minutes transiting the station. Figure 4 shows a picture looking north toward the scales and the dual lane

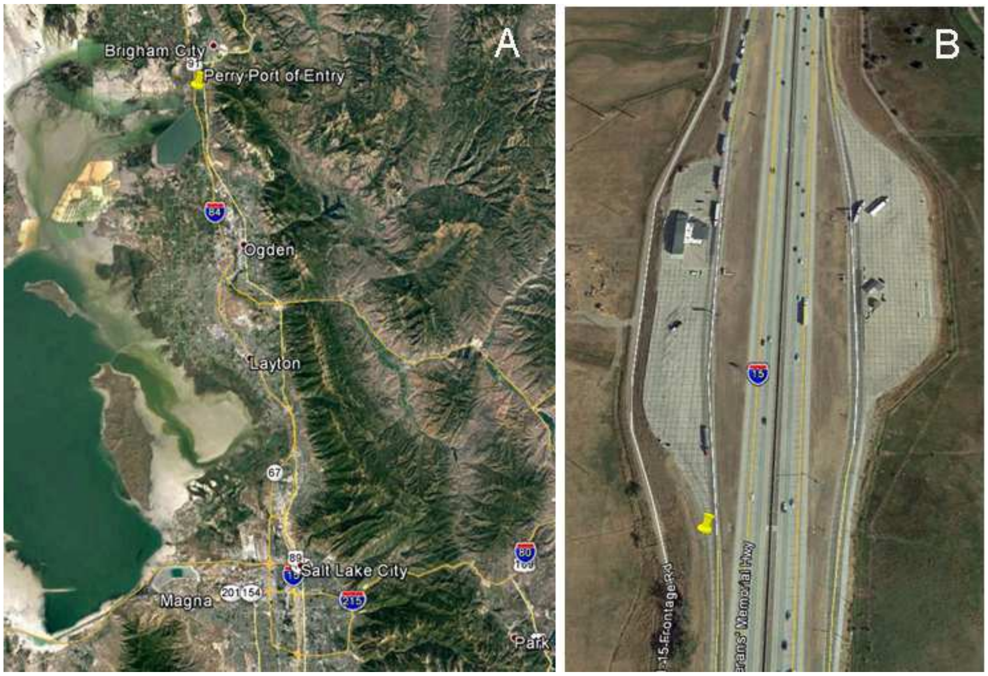

Figure 3. A satellite photo on the left (A) shows the location of the Perry Port of Entry north of the Salt Lake City area on Interstate 15. The satellite photo on the right provides a close up view of the layout of the Perry Port of Entry with approximate measurement location indicated by the yellow pin.

Figure 4. View looking north toward the scales showing the Port of Entries lane arrangement. Speed limits through the port were 3mph on the inside lane and 20 mph on the outside lane.

layout.

The majority of HDV traveling by the site are not required to enter the Port and many companies pay a fee to avoid having to stop at the station. This does not completely exclude these vehicles as any anomalies between the registered weights and the measured weights will trigger an inspection requirement and bring the vehicle through the Port of Entry. In addition HDV are selected at random for inspection and this will include some of these vehicles. It is also true that the probability of observing a particular model year in the Port is proportional to its observed frequency on the interstate. However, care should be taken when applying the age distribution observed in our sampling campaign as it may not accurately reflect the fleet using this interstate system segment.

On-road heavy-duty vehicles (HDV) manufactured since 2011 have had the option for their exhaust to exit the truck via either a traditional elevated pipe mounted to the side of the tractor or with a ground level pipe mounted underneath. This required two arrangements of the remote sensing equipment to successfully measure both type of exhaust pipe arrangements: 1) the equipment was placed atop two scaffolding towers erected on opposite sides of the travel lane to lift the remote sensors sampling beam 4.3 m above the ground to clear the tops of the passing trucks and 2) a ground level installation that allowed for sampling underneath the truck's trailer. Figure 5 shows the high installation with the two scaffolding towers with the sensors and light source mounted on top and the motor home that housed the computers and support equipment. The scaffolding was stabilized with three wires arranged in a Y shape. Figure 6 shows a corresponding picture for the ground level measurement setup. This setup also allows for the emissions of medium-duty vehicles (MDV) to also be collected. More visible in this photograph are the twin sensors used for measuring the vehicles speed and acceleration. These sensors on the near road side were attached directly to the scaffolding in the elevated setup.

The two setups also had slightly different operational software to accomplish the data collection. The High FEAT measurement was triggered with the use of an additional infrared sensors that were installed on tripods forward of the scaffolding. Interruption of this light beam initiated a 1 second exhaust measurement with data collected on each detector channel at 100 Hz. For the elevated measurements we allow for a longer window of opportunity to find the exhaust plume to allow for the different tractor lengths and exhaust pipe placements found in the fleet. The Low FEAT was triggered conventionally when a vehicle's tire passed through the Low FEAT IR beam, causing the reference signal to be blocked, and half a second of data was collected at 100 Hz for each measurement. The Low FEAT uses a shorter sampling time in order to complete the sampling before the rear trailer wheels interrupt the measurement.

Exhaust thermographs were taken with an infrared camera (Thermovision A20, FLIR Systems) for qualitatively estimating the exhaust temperatures of the trucks with elevated exhaust pipes leaving the weigh station. Thermal imaging was only attempted on trucks with elevated exhaust

Figure 5. A photograph showing the scaffolding setup for measuring exhaust from trucks with elevated pipes.

Figure 6. A photograph showing the ground level setup and the twin tripod mounted sensors used for measuring vehicle speed and acceleration.

pipes because of the difficulty acquiring a clear image of exhaust pipes from trucks with ground level exhaust. The IR camera system was capable of imaging the exhaust systems for many of the trucks that had elevated exhaust systems, and a field-calibration of this IR camera allows for these images to be converted into temperatures.[28] This is necessarily only an estimate because it is often difficult to distinguish a truly cold pipe from one that has been shielded. In addition as mention we are unable to image the ground level exhaust pipes because of their location and this restricts the number of newer model year trucks we can observe. Figure 7 shows a sample picture of a truck leaving the Port where the pipe is clearly visible and from which we were able to estimate an exhaust temperature (115 °C).[29] Outside air temperatures were recorded at the site at 5 minute intervals with the use of an Elitech model RC-5+ recording thermometer.

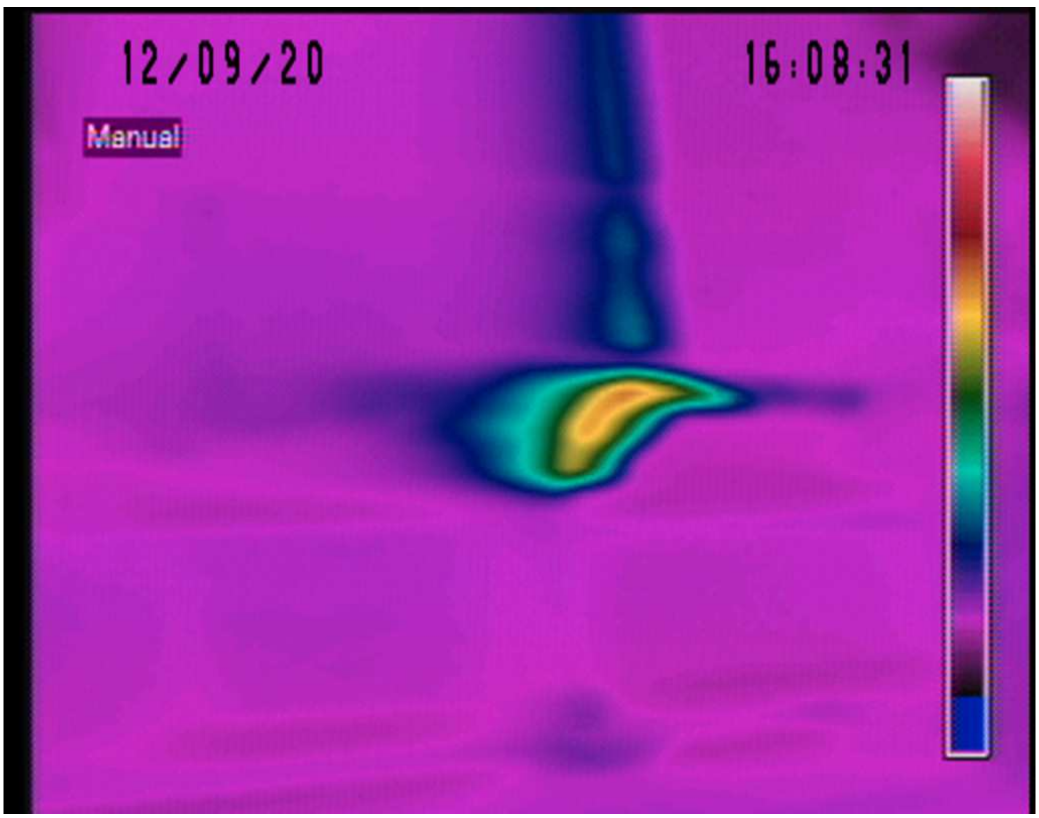

Figure 7. A thermal image of a truck with an elevated exhaust pipe collected on December 9th. Pipe temperature was estimated at 115° C for this truck.

Trucks with at least a valid CO measurement had their license plates and state and/or country manually transcribed. License plates for the states of California, Colorado, Idaho and Utah were matched against state registration records for non-personal vehicle information such as make, chassis model year and Vehicle Identification Number (VIN). The remaining license plates, and any plates that could not be matched by the four states, were manually matched using publically available registration data found online. All of the matched registration information was visually verified for vehicle make to eliminate, where possible, mismatched registration information. VIN's for the matched trucks were decoded using the National Highway Traffic and Safety Administration's online VIN decoder (https://vpic.nhtsa.dot.gov/api/) that provided additional vehicle information such as truck model, engine size, engine model, engine manufacturer, engine horse power, fuel type and weight class.

Heavy and many medium-duty trucks in the United States have emission regulations that are enforced based on the year that the engine is manufactured, not when it is installed in a chassis. Chassis model year information acquired from State motor vehicle registration records or VIN's is the year that the vehicle was assembled. It is not possible for us to unequivocally determine a vehicles engine model year as that would require an inspection of every vehicles engine sticker but past experience has shown that chassis model year on average is one model year newer than the engine model year.[10, 30] As such this report uses chassis model year as obtained from registration records and subdivides engine certification standards assuming that the chassis model year is one year older than the engine model year. So for example 2010 engine model year regulations are expected to show up in 2011 chassis model year vehicles.

Results and Discussion

Measurements were collected at the southbound Perry Port of Entry for ~32 hours over a six day period from Sunday December 6 to Friday December 11. The majority of measurements were collected during daylight hours (~28 hrs), however, on two evenings some measurements were collected after dark. The University of Denver FEAT remote sensor was operated in the elevated configuration for ~22 hours and in the ground level configuration for ~10 hours.

The 2020 Perry Port of Entry campaign resulted in 1694 measurements from HDV (1591) and MDV (103). The two vehicle classifications used for this report have been separated by gross vehicle weight > 26001 lbs. for HDV (class 7 & 8) and <26000 lbs. for MDV. Matched licenses for unique HDVs and MDVs by state and Canada are shown in Table 1. Vehicles from 37 different states were sampled with the largest numbers from Utah (35.5%) and Idaho (13.9%). Table 2 provides a summary of fleet emission averages for the High and Low setups as well as for the entire HDV and MDV fleets. The mean molar emission ratios to CO_2 are shown as well as mean and median g/kg of fuel emissions for CO, HC, NO, NO_2, NO_x, NH_3, IR %opacity, average chassis model year, speed (mph), acceleration (mph/s), vehicle specific power (VSP), the road slope (degrees) and mean outside air temperatures measured at the site (°C). Uncertainties for the 'All HDV' and 'All MDV' are standard error of the mean determined using the daily measurements.

As previously mentioned HDV equipped with ground level exhaust are a recent development and as such are generally only found on vehicles manufactured since 2011. This age range allows for a high percentage of these vehicles to be powered by engines with NO_x after-treatment systems.

Table 1. 2020 Perry Port of Entry matched license plates for HDV and MDV.

State	HDV Total (Unique)	MDV Total (Unique)
AL	3	0
AZ	8	0
CA	41	1
CO	11	1
FL	10	0
GA	6	0
IA	11	0
ID	217 (196)	11 (10)
IL	63	0
IN	163 (151)	2
KS	2	0
MD	2	0
MI	6	0
MN	19	0
MO	25	0
MS	1	0
MT	13	1
NC	6	0
ND	10	1
NE	33	0
NJ	2	0
NM	6	0
NV	3	0
NY	4	0
OH	16	1
OK	15	0
OR	48 (47)	0
PA	1	0
RI	1	0
SC	1	0
SD	2	0
TN	15 (14)	0
TX	49 (48)	5
UT	645 (501)	79 (77)
WA	45	1
WI	7	0
WY	8	0
Canada	73	0
Totals	1591 (1411)	103 (100)

Table 2. 2020 Perry Port of Entry data summary.

FEAT Number of Measurements	High HDV 1053	Low HDV 538	All HDV 1591	All MDV 103
Mean CO/CO_2 (gCO/kg of fuel)	0.004 (7.0)	0.002 (3.4)	0.003 (5.8 ± 1.5)	0.006 (10.7 ± 6.8)
Median gCO/kg	4.1	2.7	3.6	8.6
Mean HC/CO_2 (gHC/kg of fuel)	-0.0004 (-2.5)	0.0008 (4.6)	0.00001 (-0.08 ± 0.07)	0.0008 (4.7 ± 5.1)
Median gHC/kg	-1.3	1.7	-0.04	3.9
Mean NO/CO_2 (gNO/kg of fuel)	0.0066 (14.2)	0.0029 (6.2)	0.0054 (11.5 ± 1.3)	0.0043 (9.1 ± 4.9)
Median gNO/kg	8.3	2.2	5.3	2.5
Mean NH_3/CO_2 (gNH_3/kg of fuel)	0.00001 (0.009)	0.0002 (0.23)	0.00007 (0.08 ± 0.06)	0.0002 (0.22 ± 0.23)
Median gNH_3/kg	-0.01	0.07	0.01	0.02
Mean NO_2/CO_2 (gNO_2/kg of fuel)	0.0002 (0.72)	0.0002 (0.56)	0.0002 (0.67 ± 0.09)	0.0002 (0.77 ± 0.3)
Median gNO_2/kg	0.39	0.14	0.28	0.34
Mean gNO_x/kg Median gNO_x/kg	22.5 13.5	9.9 3.4	18.5 ± 2.0 8.9	14.5 ± 7.8 4.7
Mean IR %Opacity Median IR %Opacity	0.7 ± 0.1 0.5	0.4 0.3	0.6 ± 0.1 0.5	0.6 0.7
Mean Chassis Model Year	2012.6	2017.4	2014.2	2015.2
Mean Speed (mph)	27.9	30.9	28.9	32.8
Mean Acceleration (mph/s)	0.3	0.01	0.2	-0.4
Mean STP(skw/tonne) Slope (degrees)	6.8 0°	4.4 0°	6.0 0°	4.6 0°
Mean Temperature (°C)	3.9	3.6	3.8	3.8

A comparison between the fuel specific NO_x emissions for High HDV and Low HDV measurement sets, shown in Table 2, show that the emissions are 56% lower for the HDV measured with ground level exhaust. However, that vehicle grouping is also approximately five years newer on average (2017.4 vs 2012.6) than the High HDV data set, reflecting the higher percentage of trucks equipped with NO_x after-treatment systems. The ground level setup also captured a small number of MDV (103). The limited size of this group coupled with it being a mixture of gasoline and diesel powered vehicles contributes to the large uncertainties observed for those emission measurements and the results from these vehicles will not be discussed in any more detail.

Utah plated vehicles make up the largest percentage (35.5%) of the vehicles observed at the Port. Table 3 is a data summary for only HDV comparing the measurements for the Utah and Out of State fleet. The Utah plated fleet is 2.4 years older than the Out of State fleet and most mean emission values are higher as well. Figure 8 compares the fleet percent by model year for these two fleets. The Out of State fleet is characterized by a higher percentage of 3 year old and newer trucks (49% versus 36%) indicative of the influence of long-haul trucks, while the Utah fleet has almost double (31% versus 16%) the number of HDV model year 2010 and older. The large percentage of 2006 and 2007 trucks reflects the national purchase trend for these model years as they preceded the introduction of the required installation of diesel particulate filters.

Compression ignition engines are operated with excess air which generally leads to lower operational levels of CO and HC emissions. Figures 9 and 10 graph the fuel specific CO and HC emissions by chassis model year for the HDV measured at the Perry Port of Entry. The uncertainties plotted are standard error of the mean determined from the daily measurements. In general HDV with selective catalytic reduction (SCR) systems operate with high air to fuel ratios to maximize fuel economy and allow the SCR to reduce the NO_x emissions. However, prior to SCR systems, NO_x emissions were often reduced by recirculating exhaust gases (EGR) into the engine cylinders thus lowering the air to fuel ratio which most often increased CO and particle emissions. This was especially true for the 2008 - 2010 chassis model year vehicles when diesel particulate filters (DPF) where first installed as manufacturers could rely on the DPF to control the particle emissions. In general the CO emissions increase as the age of the vehicles increase but the large uncertainties hide some of these differences. If we group the HDV into post-2010 (3.7 ± 0.9 gCO/kg of Fuel) and pre-2011(12.9 ± 3.6 gCO/kg of Fuel) models the differences in engine operations is easier to see. These differences are mirrored in the infrared opacity measurements collected as well with the post-2010 (0.51 ± 0.05 % IR Opacity) also showing lower soot emissions than the pre-2011 (0.84 ± 0.14 % IR Opacity) models. HC emissions are very low and generally scatter around zero.

Figure 11 is a bar chart for the mean fuel specific NO_x emissions by model year for the class 7 and 8 heavy-duty trucks measured at the Perry Port of Entry. Each bar is apportioned with the solid portion of each bar showing the contribution to the total NO_x emissions from NO and the

Table 3. 2020 Perry Port of Entry data comparison for Utah and Out of State HDV.

FEAT Number of Measurements	Utah HDV 645	Out of State HDV 946
Mean CO/CO_2 (gCO/kg of fuel)	0.003 (6.5 ± 1.8)	0.003 (5.2 ± 1.3)
Median gCO/kg	3.7	3.5
Mean HC/CO_2 (gHC/kg of fuel)	0.000003 (-0.2 ± 0.3)	0.0003 (-0.002 ± 0.002)
Median gHC/kg	-0.1	0.08
Mean NO/CO_2 (gNO/kg of fuel)	0.0057 (12.2 ± 1.7)	0.0051 (11.0 ± 1.2)
Median gNO/kg	6.5	4.6
Mean NH_3/CO_2 (gNH_3/kg of fuel)	0.00002 (0.02 ± 0.03)	0.0001 (0.12 ± 0.08)
Median gNH_3/kg	-0.008	0.02
Mean NO_2/CO_2 (gNO_2/kg of fuel)	0.0002 (0.65 ± 0.12)	0.0002 (0.68 ± 0.09)
Median gNO_2/kg	0.32	0.27
Mean gNO_x/kg Median gNO_x/kg	19.8 ± 2.8 10.8	17.5 ± 1.9 7.7
Mean IR %Opacity Median IR %Opacity	0.75 ± 0.07 0.6	0.48 ± 0.07 0.4
Mean Chassis Model Year	2012.8	2015.2
Mean Speed (mph)	28.6	29.1
Mean Acceleration (mph/s)	0.1	-0.3
Mean STP(skw/tonne) Slope (degrees)	5.4 0°	6.5 0°
Mean Temperature (°C)	3.9	4.1

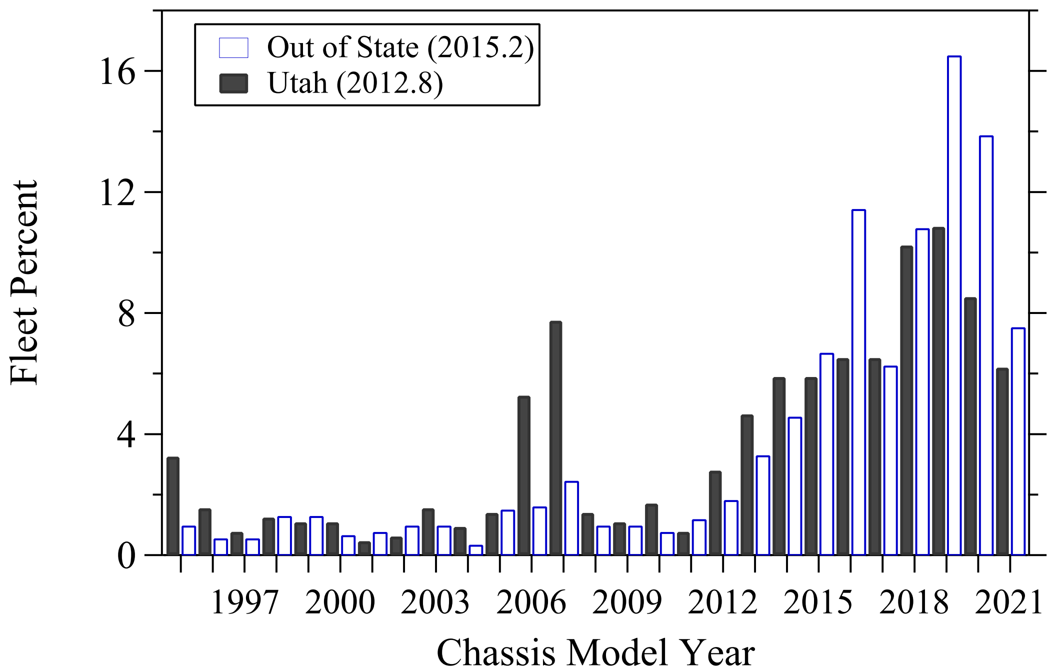

Figure 8. Fleet percent versus model year for the Utah plated HDV fleet and the Out of State HDV fleet.

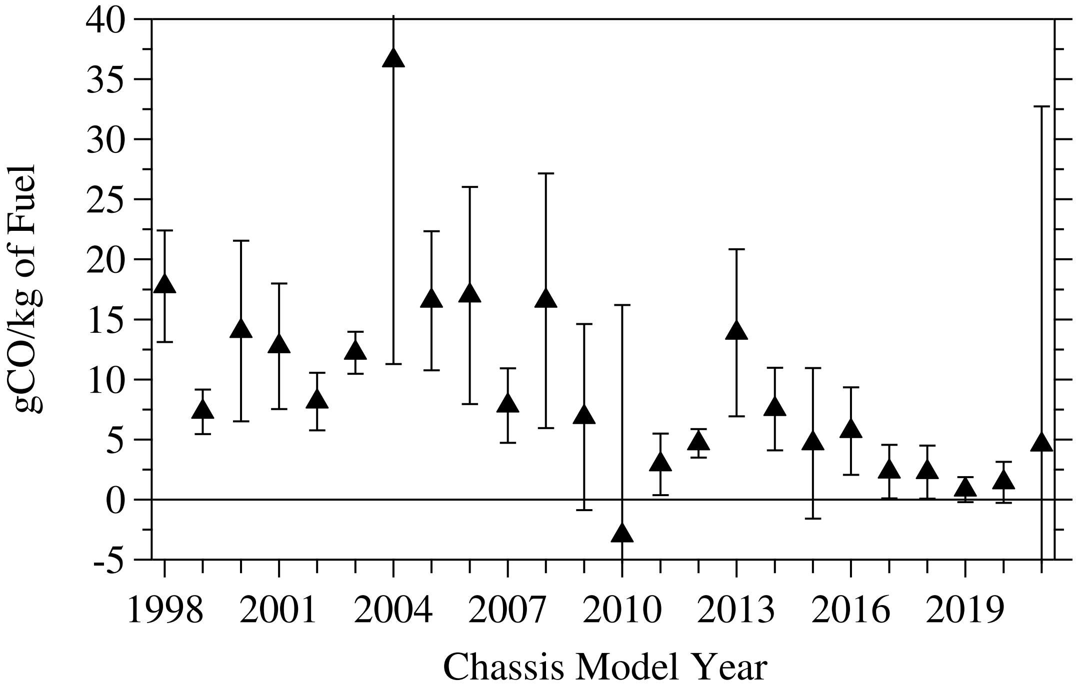

Figure 9. Fuel specific CO emissions by model year for all the HDV measured. Uncertainties are standard error of the mean calculated from the daily means.

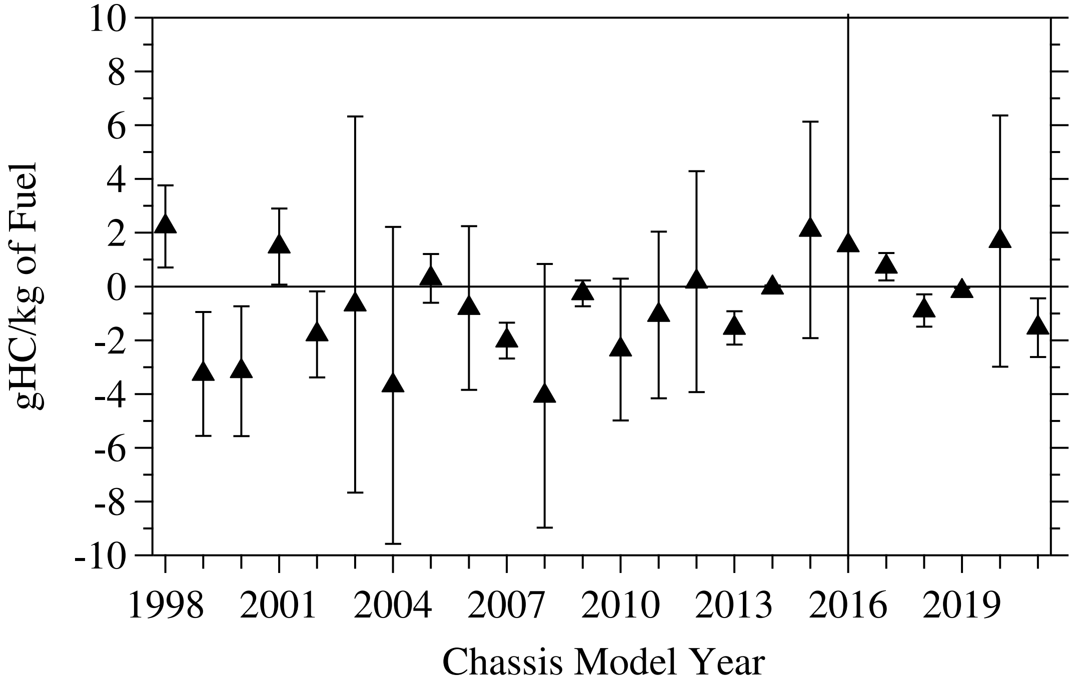

Figure 10. Fuel specific HC emissions by model year for all the HDV measured. Uncertainties are standard error of the mean calculated from the daily means.

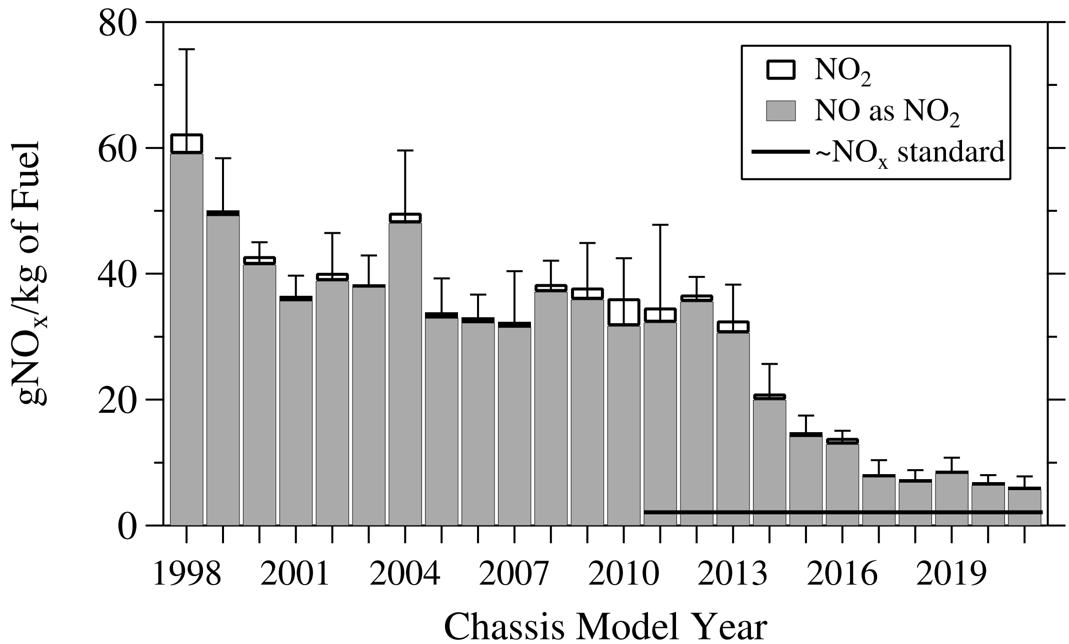

Figure 11. Total gNO_x/kg of fuel (total bar height) for HDV. Mean gNO_2/kg of fuel (open) and gNO/kg of fuel as gNO2/kg of fuel (solid) by chassis model year. Uncertainties are standard error of the mean calculated from the daily means. The NO_x standard assumes an on-road enforcement limit of 0.35 gNO_x/bhp-hr and 0.15 kg of fuel/bhp-hr (~2.1 gNO_x/kg of fuel).

open portion of each bar the contribution from NO_2. The uncertainties plotted are standard error of the mean determined from the daily measurements. The solid black line drawn between model years 2011 and 2021 represents the homologous NO_x certification standard derived from the on-road enforcement limit of 0.35 grams NO_x/brake-horsepower hour and assuming 0.15 kg of fuel is consumed per brake-horsepower hour. Keep in mind that not all of the pre-2017 HDV are certified to this low emission standard and the line only applies to the HDV that were certified to the 0.2gNO_x/bhp-hr standard.

The HDV NO_x emissions by model year trends observed at the Utah site are distinguished by two relatively stable regions of emissions linked with a short transition between the two. 2017 and newer model year HDV form the first group with the lowest NO_x emissions that within the measurement uncertainties are all similar indicating little to no emissions deterioration on average over the five years. The NO_x emission increase between the 2016 and 2013 model year vehicles to a second and significantly higher NO_x emission levels where again within the measurement uncertainties there are no real differences from the 2013 to the 2000 chassis model year HDV. The NO_2 contribution at the tailpipe is small as shown with the majority of the NO_x emissions contributed by the engine out NO emissions.

As previously mentioned we do not have any additional HDV emission measurements collected during the winter months. Our most recent HDV measurements were collected in the spring of 2017 at a weigh station (Peralta, elev. 104m) on CA-91 in the Anaheim Hills in the South Coast Air Basin in California. Figure 12 graphs the fuel specific NO_x emissions by chassis model year comparing the 2020 Utah and 2017 California HDV measurements. The uncertainties are standard error of the mean determined from the daily measurements. The differences in this comparison are obvious as the California measurements show a significantly slower increase in NO_x emissions between the newest and oldest HDV. In addition NO_x emissions observed in Utah are also higher at both ends of the age distribution as well.

However, there are many differences between these two fleets and measurements that need to be factored in to fully understand the comparison. The first is the HDV measured in Utah are 3.5 years older than the same chassis model year vehicle measured in California. Past research campaigns at this same California weigh station has shown a pattern of fuel specific NO_x emissions deterioration with emission increases with increasing vehicle age.[31] Figure 13 duplicates the 2020 Utah and 2017 California data shown in the previous figure (Figure 12) but adds measurements collected at the same location in 2012. It is easy to see that fuel specific NO_x emissions are higher for each of the similar chassis model year observed during the five year interval showing the negative effect of age on HDV NO_x emissions.

To discuss some of the other factors likely involved in the differences we are going to look beyond the mean emissions and look at differences in the NO_x emissions distribution between the two sites. Figure 14 is a box and whisker plot comparing the fuel specific NO_x emissions by

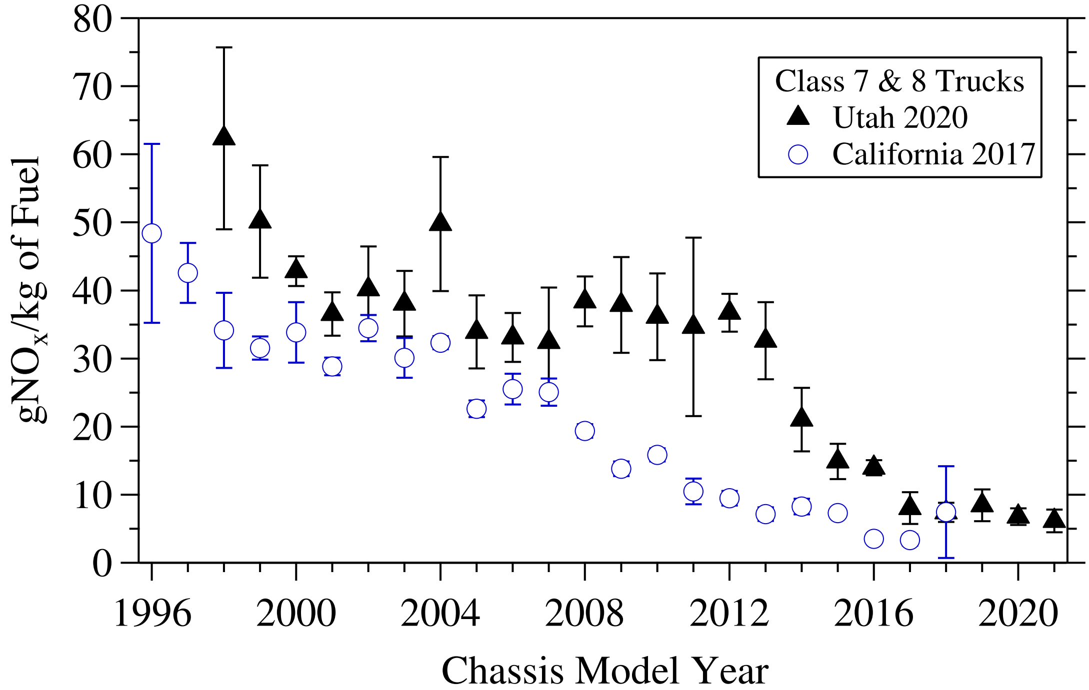

Figure 12. HDV gNO_x/kg of fuel versus chassis model year for the 2020 Utah measurements (triangles) and the 2017 California measurements (circles). Uncertainties are standard error of the mean calculated from the daily means.

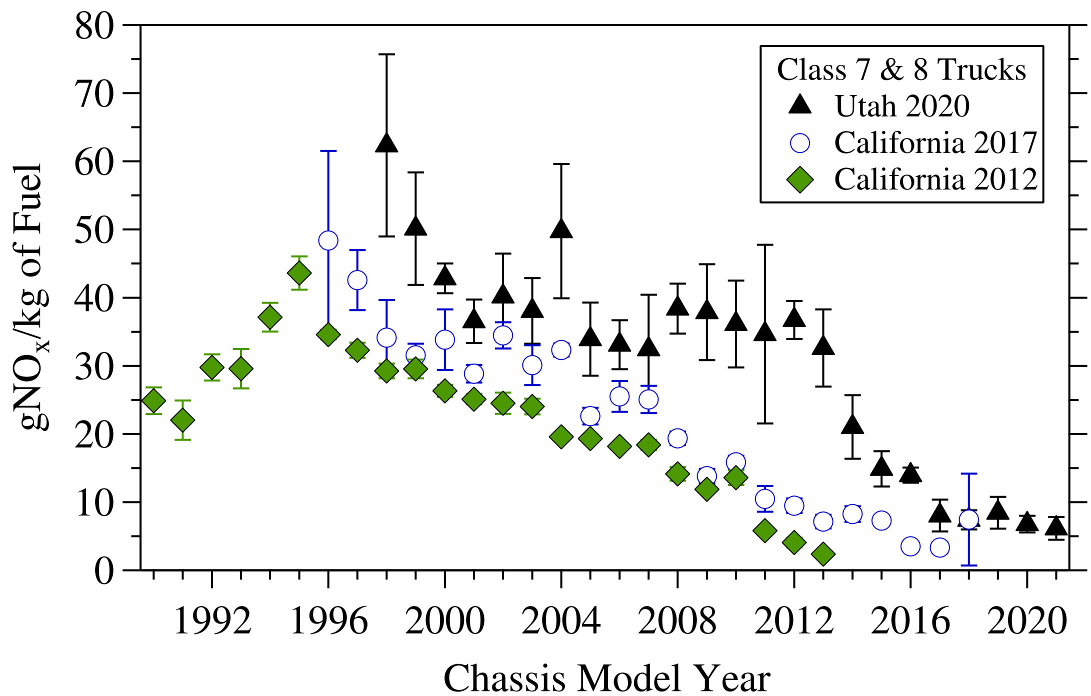

Figure 13. HDV gNO_x/kg of fuel versus chassis model year for the 2020 Utah measurements (triangles) the 2017 (circles) and 2012 (diamonds) California measurements. Uncertainties are standard error of the mean calculated from the daily means.

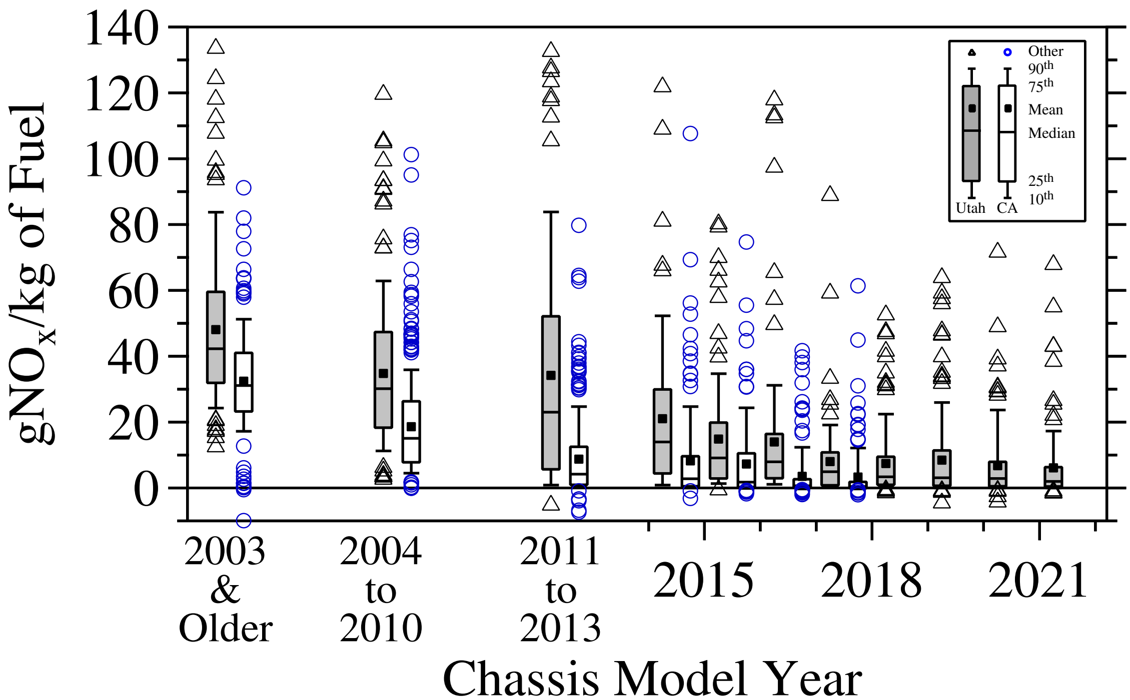

Figure 14. Box and whisker plot comparing the fuel specific NO_x emissions by chassis model year for HDV measured in 2020 in Utah and in 2017 in California. The box defines the 25^{th}, 50^{th}, and 75^{th} percentiles with the whiskers extending from the 10^{th} to the 90^{th} percentile. The symbols for each location are measurements that lie beyond 1.25 times the whisker length from the median. The mean for each group is plotted as the solid square.

chassis model year for HDV from the 2020 Utah and 2017 California measurements. For chassis model years older than 2014, we have grouped together multiple model years in order to increase the number of measurements in each group. The box defines the 25^{th}, 50^{th}, and 75^{th} percentiles with the whiskers extending from the 10^{th} to the 90^{th} percentile. The symbols for each location are measurements that lie beyond 1.25 times the whisker length from the median. The mean emissions for each group are plotted as filled squares. It is obvious again that not just the mean emissions are larger for the Utah data set but the spread of the emissions distribution is also larger especially for the 2013 and older chassis model year groupings.

Environmental Effects on NO_x Emissions. Environmental factors of elevation and temperature are also different between these two sites. The California weigh station is located at an elevation of approximately 108m while the Perry Port of Entry is a little more than 1km higher in elevation at 1300m. However, it should be pointed out that the Utah location is still under the elevation limit of approximately 1676m that manufacturers are required to certify emissions performance in the U.S. The different seasons that the measurement campaigns were conducted also leads to a temperature difference. The March California measurements were collected with a temperature range of 15.5 - 20°C while the Utah December campaign saw a range of -7 to 10°C a 22.5 to 10°C temperature difference between the low and high extremes. These temperature differences are evident in the IR thermographs collected at the two sites as well. Figure 15 is bar chart

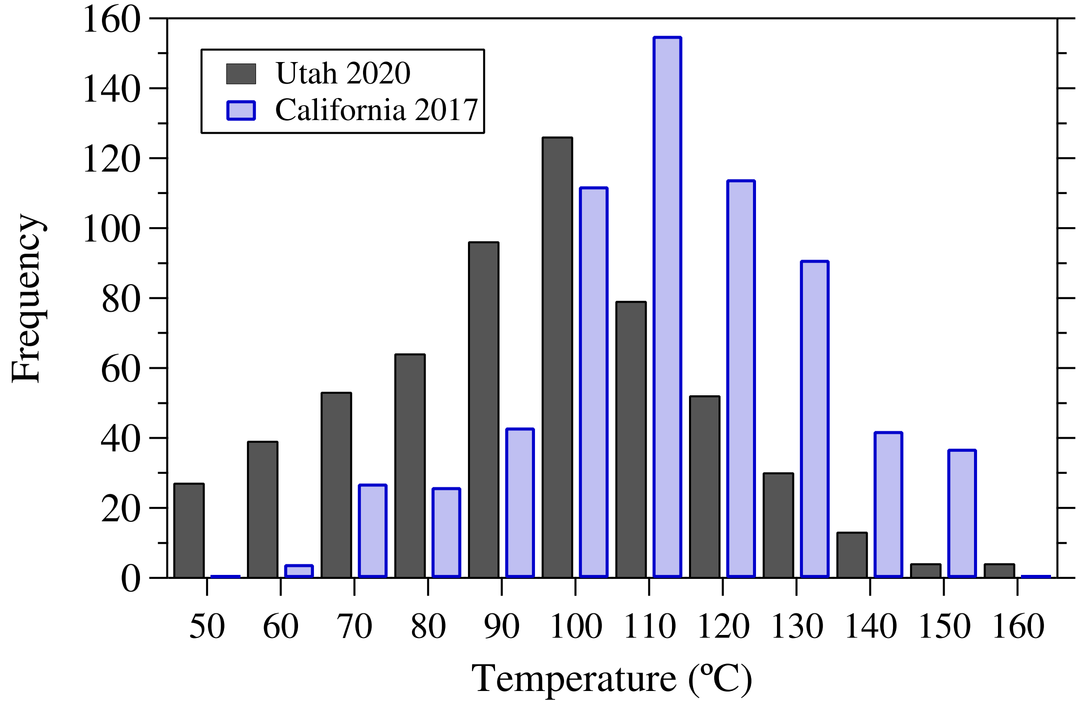

Figure 15. Comparison of the temperature distribution observed from elevated exhaust pipes during the winter 2020 Utah measurements and the spring 2017 California measurements.

comparing the temperature distribution of the elevated HDV pipe temperatures recorded at the Utah and California sites. For this work not only were the ambient temperatures lower but we found that the pipe temperatures were lower as well with an estimated mean pipe temperature of 92° in Utah compared to 110° reading from the California study.

Unequivocally separating these individual influences is likely not possible with a single data set but because of the differences in engine management systems (mechanically versus electronically controlled) found in the Utah fleet it does allow us to estimate the possible magnitude of the effects on vehicle NO_x emissions. 2003 and older HDV were manufactured with mechanically controlled engines that were not designed to fully compensate for changes in altitude and temperature during operation. The box and whisker plot in Figure 14 shows a NO_x mean emission difference of 32.6% between the Utah (48.09 ± 2.7 gNO_x/kg of fuel) and the California (32.4 ± 0.2 gNO_x/kg of fuel) 2003 and older vehicles. If we normalize the model year distribution between the two fleets it lowers the Utah mean to 45.2.

Research has shown that altitude can increase NO_x emissions in mechanically controlled engines ~6.3 gNO_x/kg of fuel/km increase which is about half of the difference observed in the 2003 and older HDV.[32] This would leave temperature and aging effects to possibly account for the remaining differences. The age differences between the two fleets (Utah is 3.5 years older) could undoubtedly account for all of the remaining difference. However, even if the remaining emissions were equally the result of age and temperature affects that limits the temperature to only a minor influence of less than 10% (~3.2 / 45.2 gNO_x/kg of fuel).

At the other extreme are the newest fully electronically controlled engines found in the 2011 and newer HDVs. If however, we focus only on the 3 year old and newer trucks shown in Figure 14 with the associated age differences between the two sites this will select 2018 - 2021 chassis model year vehicles in Utah and 2014 - 2017 chassis model year vehicles in California. What is noticeable is that the interquartile range for these vehicles are closer in size between the two sites than for the older chassis model year vehicles. Additionally, despite the environmental differences there are NO_x emission outliers that extend beyond the 90th percentile to similar emission levels. Table 4 lists the mean fuel specific NO_x emissions for each sites 4 model years.

Table 4. Fuel Specific NO_x Emissions Comparison for 3 Year Old and Newer HDV.

Utah 2020		California 2017	
Chassis Model Year (Age)	Mean gNO_x/kg of Fuel ± SEM	Chassis Model Year (Age)	Mean gNO_x/kg of Fuel ± SEM
2021 (0)	6.12 ± 1.67	2017 (0)	3.31 ± 0.80
2020 (1)	6.78 ± 1.21	2016 (1)	3.51 ± 0.22
2019 (2)	8.43 ± 2.33	2015 (2)	7.28 ± 0.28
2018 (3)	7.39 ± 1.41	2014 (3)	8.25 ± 1.17
Overall Mean	7.19 ± 0.97	Overall Mean	5.34 ± 0.31

Within the uncertainties, all of the Utah chassis model year 2018 - 2021 mean NO_x values are similar. The differences found when comparing Utah and California 0 to 3 year old vehicles is that the first two model year in California have significantly lower emissions which account for the overall mean emissions differences (~25%). Comparing both sites means to the on-road Federal enforcement standard of ~2.1 gNO_x/kg of Fuel (assuming 0.35 gNO_x/bhp-hr and 0.15 kg of fuel/bhp-hr) show that even the California HDV still on average exceed this threshold, though the 2016 - 2017 chassis model year vehicles are very close within the uncertainties.

Temperature should not in general change the operation of these newer engines and their fuel management; however, it could affect the operation of the NO_x after-treatment systems. In a modern diesel HDV after-treatment systems that are downstream of the engine typically require an operating temperature in excess of 150°C for operation of the SCR. However, the majority of SCR systems on the road (copper zeolite substrates) have a very steep NO_x conversion efficiency curve, shown in Figure 16, that starts around a 10% conversion efficiency at 150°C and approaches 90% conversion efficiency above 200°C.[33] Temperature is not only required for proper SCR function but the thermalization of aqueous urea to NH_3 is a necessary step in the process as well. Depending on the manufacturer's threshold, low after-treatment temperatures can interrupt the urea dosing, SCR function or both. In these circumstances the vehicle's NO_x emissions are not limited by any Federal emission standards and since modern diesel engines are generally maximized for fuel economy, and therefore high NO_x emissions, during periods that

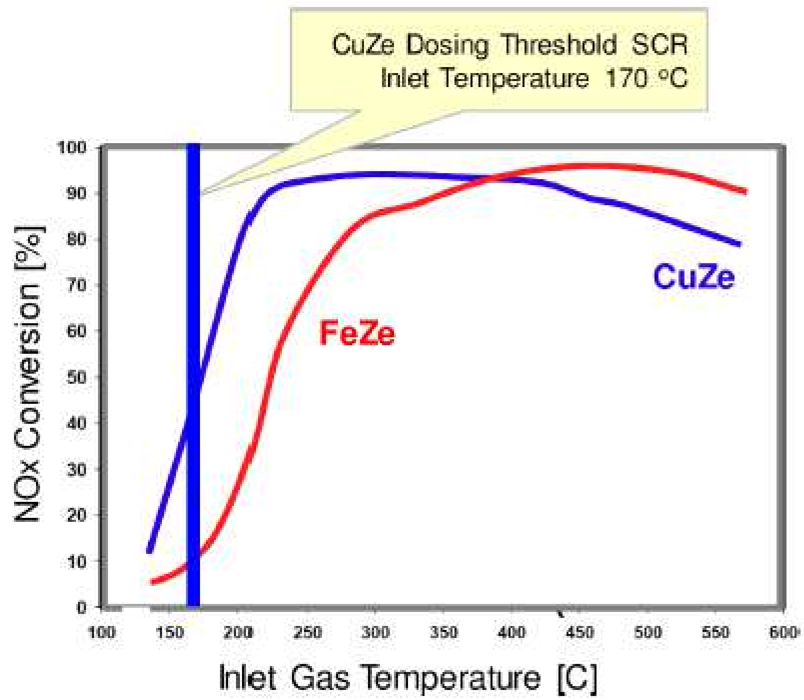

Figure 16. NO_x conversion versus SCR inlet gas temperature for two substrate types of SCR catalyst materials (Stanton, 2013).

the after-treatment system is not fully operational can lead to temporarily high levels of NO_x emissions.

Regardless of the outside temperature when a HDV exits the highway to transit the inspection station the slower speeds provide time for the after-treatment systems to cool and be totally or partially offline during the acceleration event as the truck gains speed to reenter the highway. This is analogous to the driving situations encountered in freeway or arterial driving in congested urban traffic. It's important to remember that even if dosing is interrupted during this period any available NH_3 still on the SCR may be used to reduce NO_x exhaust emissions. This situation may lead to increased NO_x emissions but not completely uncontrolled emissions. The high NO_x emissions observed beyond the 90th percentile whiskers in Figure 14 in the newer model year vehicles at both sites are likely the result of some type of diminished after-treatment function that is not related to the ambient temperature.

As previously mentioned the extent and number of outliers above the 90th percentiles for these 3 year old and newer vehicles is similar between the two sampling locations despite the lower temperatures experienced in Utah. So if the lower temperatures in Utah are not directly resulting in an increase in the number of HDV observed with high NO_x emissions then where are the emission differences occurring that account for the overall increases in the mean NO_x emissions. Figure 17 is a cumulative probability plot for the Utah 2020 and California 2017 3 year old and newer HDV fuel specific NO_x emissions showing the probability of finding a specific NO_x

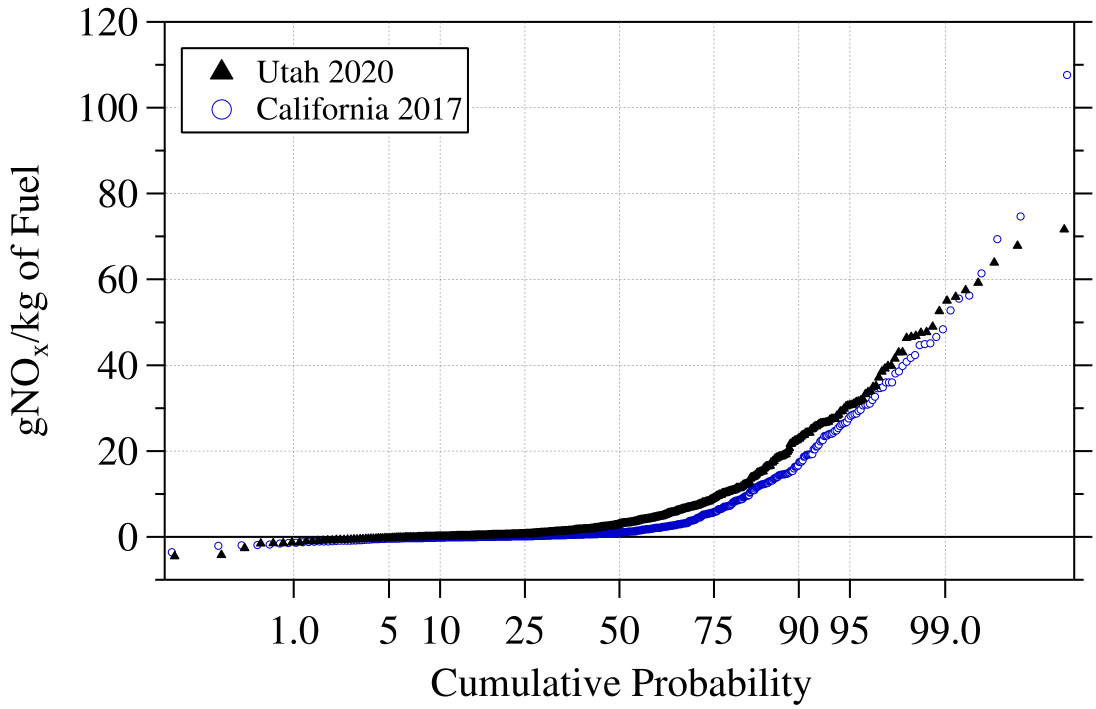

Figure 17. Cumulative probability plot of fuel specific NO_x emissions for the 3 year old and newer HDV measured in Utah and California. The x-axis has been transformed to a normal distribution. If the data sets were normally distributed they would plot as a diagonal straight line.

emissions level in the distribution. The x-axis has been transformed to a normal distribution where if the distribution was normally distributed it would plot as a diagonal straight line. A careful comparison of the two distributions shows that the Utah measurements for the newest HDV begin to rise above the California measurements between the 10th and 25th percentiles and by the median parallel the increases observed in the California data. Above the 95th percentile we start to see similar probabilities in both data sets of finding the extreme NO_x emissions.

To better show this Figure 18 is a quantile - quantile plot comparing the rank ordered fuel specific NO_x emission distributions from the 3 year old and newer HDV in the Utah and California data sets. For this plot we have calculated the quantiles for each data set from 2.5 to 99% using steps of 2.5% and the solid line is the 1:1 line. When the shapes of the two distributions are similar, the points in these plots will fall along a straight line though not necessarily along the 1:1 line. It is easier to notice that the two distributions for the lowest emission levels begin with the points falling along the 1:1 line. Slightly above zero the Utah measurements increase faster than the observations from California and they rise to a point where they level out and then parallel the 1:1 through the remainder of the quantiles. This again indicates that the high emission tails of the two distributions are distributed similarly but that the lower to the middle part of the distribution see's higher NO_x emissions in the Utah measurements. This suggests that the effect of temperature on these newest model years HDV does not work to increase the number of SCR systems that are completely deactivated but to

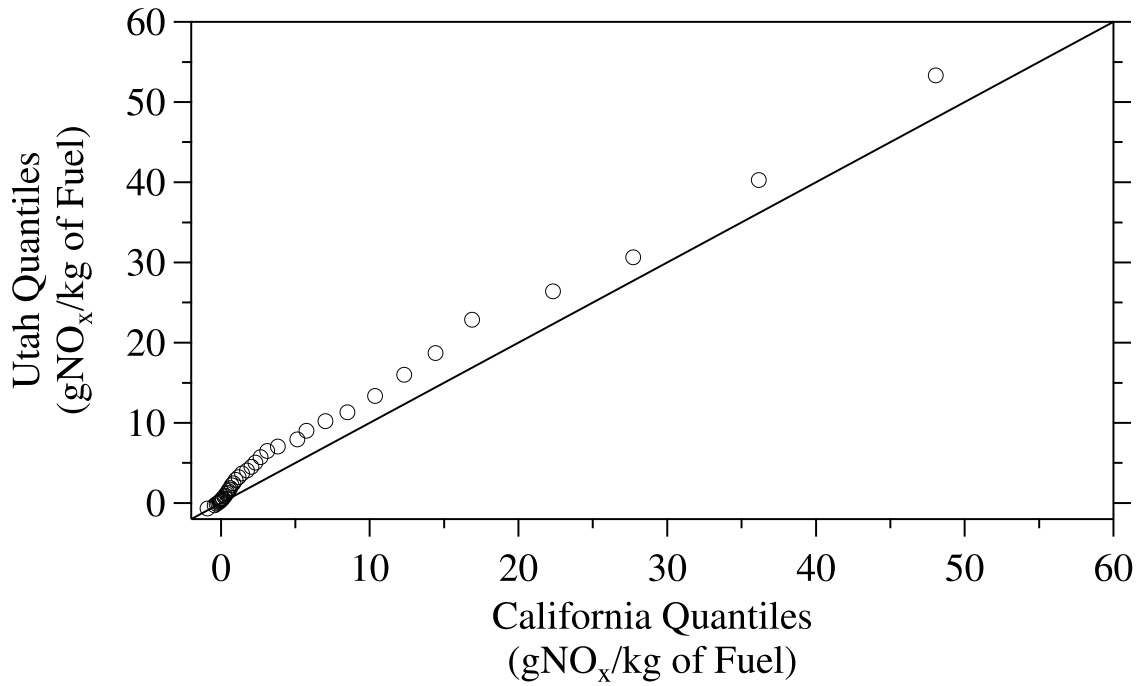

Figure 18. Quantile - Quantile plot of fuel specific NO_x emissions for the 3 year old and newer HDV measured in Utah and California comparing the emissions distribution. Quantiles range from the 2.5th to the 99th. The solid line is a 1:1 line.

lower the NO_x conversion efficiency in a significant number of vehicles resulting in the 25% increase observed in the mean NO_x emissions for this group.

Using the FLIR thermographic images we can add additional information to support the hypothesis that lower temperatures increase HDV NO_x emissions. Figure 19 is a plot of fuel specific NO_x emissions as a function of the exhaust pipe temperature in degrees Celsius for the 3 year old and newer HDV with elevated exhaust pipes for the Utah and Peralta CA measurements. Uncertainties are standard error of the mean calculated from the daily means. Because we are not able to image the ground level exhaust pipes because of their enclosed location the number of measurements that we have to compare is reduced by about 2/3 in both the Utah (194/671) and California (248/719) data sets. Nonetheless the relationship we see between the fuel specific NOx emissions and exhaust pipe temperature is similar between the two locations with the Utah relationship being offset to higher NO_x emissions. For the Utah measurements there is about a factor of 2 reductions in NO_x emissions over this temperature range. For the HDV with a thermographic pipe image there is a 10°C difference between the Utah (95°C) and the California (105°C) trucks. The temperature difference is slightly smaller than we observed (92 - 110°C) using the images for all of the HDV.

Load effects on NO_x emissions. An additional consideration to account for in the comparison with the California measurements is operational load. The U. S. Environmental Protection Agency uses scaled tractive power (STP) in their MOVES3 computer model and is a metric to

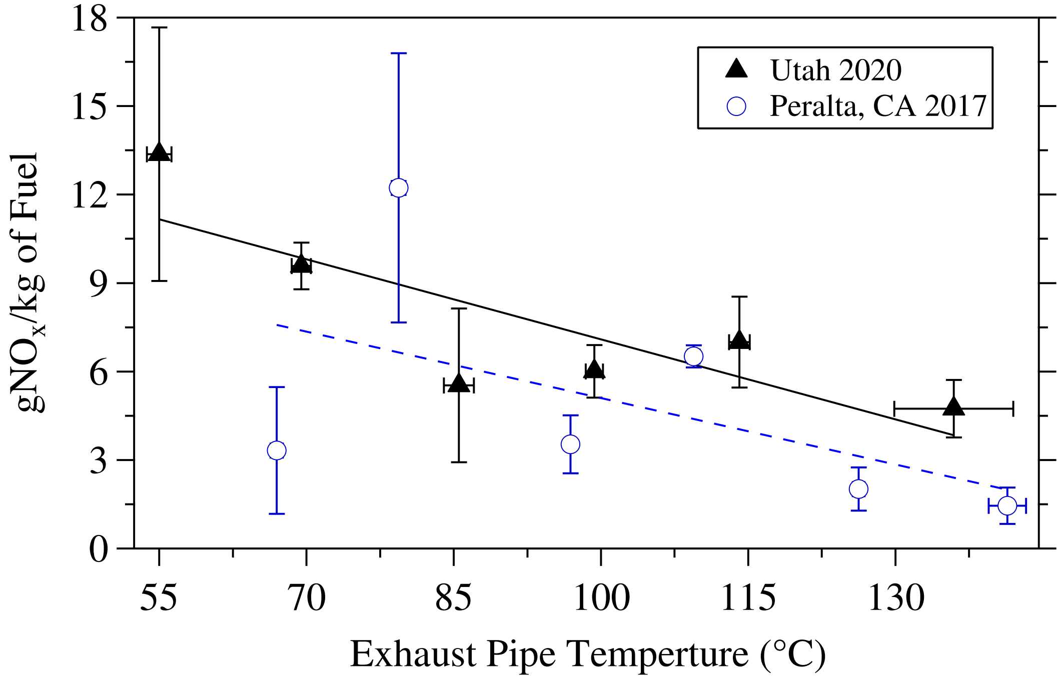

Figure 19. Fuel specific NO_x emissions versus exhaust pipe temperature (°C) for the 3 year old and newer HDV in the Utah and California data sets. The solid line is a least squares best fit line to the Utah data (slope = -0.09, R^2 = 0.68) and the dashed line is the best fit line to the Peralta CA data (slope = -0.075, R^2 = 0.28). Uncertainties are standard error of the mean determined from the daily measurements.

represent the vehicle's tractive power.[34] It is similar to vehicle specific power but uses a scaling factor to represent the average mass of a specific source type. For our application this source type was chosen to represent the class 7 and 8 combination tractor and trailer vehicles most commonly associated with HDV. STP is a function of the vehicles speed and acceleration along with the road grade which at the Perry Port of Entry was flat or 0°. STP was calculated for all of the vehicles with a valid speed and acceleration measurement which for the class 7 & 8 HDV consisted of 87% of the emission measurements.

Because of the design of the Utah and California HDV inspection stations the driving characteristics have a few differences. The California station has a rather short lead out back to the freeway that is accessed up a 1.6° uphill grade with the FEAT instrumentation setup at the beginning of the hill. This results in lower overall speeds for the HDV (14 mph) and higher accelerations (0.7 mph/sec) for a mean STP of 8.3 skw/tonne. At the Perry Port of Entry the measurement location is significantly farther away from the scales than in California allowing the HDV more time to gain speed before the measurement location. This results in the opposite situation with overall speeds significantly higher (28.9 mph) and acceleration rates lower (0.2 mph/sec) for a mean STP of 6.0 skw/tonne. Figure 20 is a bar chart showing the fleet percentage versus STP bin for the two sites detailing how the measurements are distributed. Interestingly the peak STP is the same for both locations with the 5 skw/tonne bin. However, the California

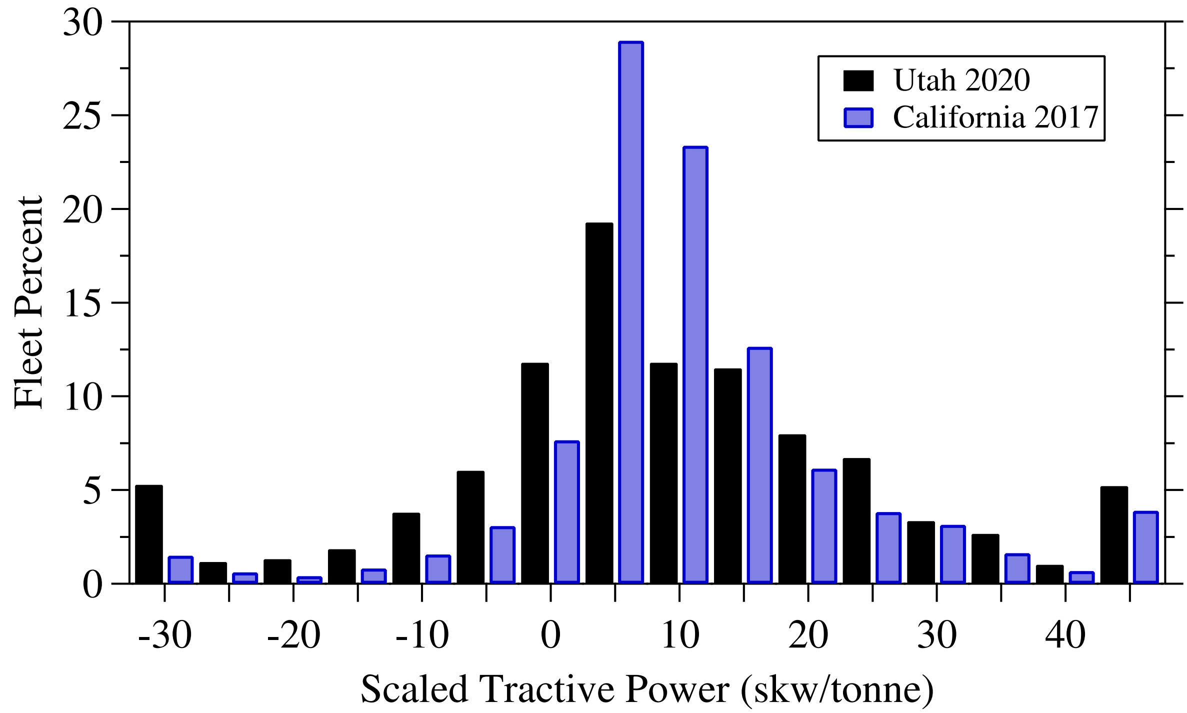

Figure 20. Fleet percent versus scaled tractive power bin (skw/tonne) for the 2020 Utah and 2017 California HDV measurements.

measurements have a majority of measurements in the 5, 10 and 15 skw/tonne bins (65% of the measurements) while those bins only account for 42% of the Utah measurements. In addition the Utah observations have a decidedly larger negative tail with the 0, -5 and -10 bins accounting for 21% of the measurements which likely indicates coasting through the measurement site.

Figure 21 graphs the fuel specific NO_x emission versus STP for the Utah and California measurements. The endpoint bins in the graph contain all measurements that are lower or higher than the respective bin. Uncertainties are standard error of the mean calculated using the daily measurements. Within the uncertainties of the measurements the NO_x emissions for both fleets exhibit little dependence in STP and are generally flat across the range plotted. The Utah fleet has higher NO_x emissions than the California fleet for all STP bins but two, -5 and the -30 bins. In general we expect to have higher NO_x emissions with higher loads but the opposite is observed in this case and suggests that load differences are not a major factor in the observed NO_x emission differences.

<u>Emission modeling comparison.</u> One of the motivations for this work is to improve the understanding of the Salt Lake City regions NO_x emissions inventory, especially during the winter season. Most regions in the U.S. rely on the Environmental Protection Agencies MOVES computer model for on and off-road vehicle emissions to include in their inventories.[35] With that in mind we have utilized the most recent revision of this model, MOVES3, to calculate the tailpipe running emissions for a Utah HDV fleet. The model was run for December 2020 using Utah Department of Transportation travel demand model data for speeds and vehicle miles

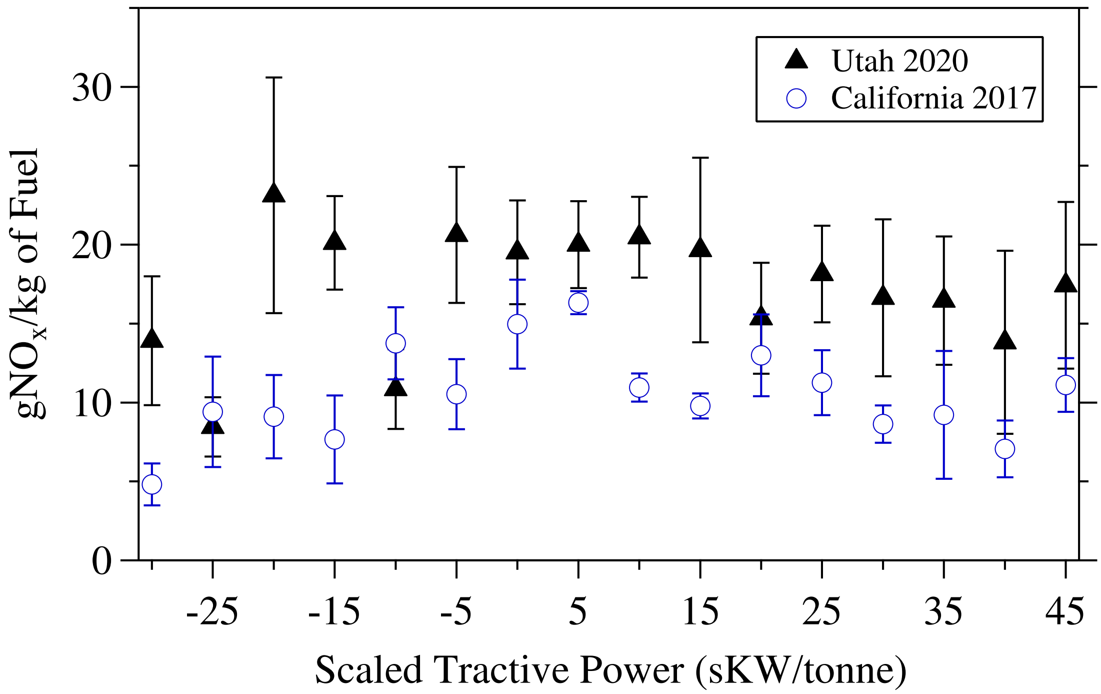

Figure 21. Fuel specific NO_x emissions versus scaled tractive power bins for the Utah and California measurements. Uncertainties are standard error of the mean calculated using the daily means. The two endpoint bins contain all measurements above or below them.

traveled and the meteorology profile from MOVES3 for Box Elder county (the county the Perry Port of Entry is located in) using the urban restricted access road type. Total running emissions (no start emissions) in grams/day were calculated for heavy-duty diesel trucks (model types 46, 47 and 49) by engine model year from 1990 to 2020. Appendix C lists model parameters and the Total Emissions output tables obtained from the MOVES3 run. Since we are only calculating running emissions the only influence temperature has in the model is through the absolute humidity correction factor. In addition MOVES3 emission values for HDV are output for engine model year and we have assumed that chassis model year (what we obtain from registration records) is one year newer than the engine model year. Molar ratios were calculated to match the FEAT measurements by converting the grams/day emissions into moles/day and then ratioing the individual species (CO, THC, NO and NO_x) to moles of CO_2. Fuel specific emission in gram of pollutant/kg of Fuel burned were calculated from the molar ratios using the same equations used for the FEAT measurements assuming 12 gCarbon/mole and 860 gCarbon/kg of fuel.[36]

For an additional comparison we also utilized California's EMFAC2021 vehicle emissions model to compute a similar set of running emissions (CO, Total Organic Gases, NO_x and NH_3) by model year.[37] A statewide region was selected for a winter season with aggregated speeds for all of the EMFAC2021 truck types with a gross vehicle weight > 26,000 lbs and diesel fuel. EMFAC2021 outputs running emissions in short tons/day (2000 lbs to the ton) and fuel consumption in 1000 gallons/day for 2001 to 2021 chassis model year vehicles. Emissions data ends with the 2001 chassis model year because of the California Truck and Bus rule which has forced the early retirement of on-road HDV older than 2001.[7] Emissions in tons/day are converted into grams/day and gallons of fuel/day are converted into kg of fuel/day assuming Ultra Low Sulphur Diesel fuel has a density of 3.255kg/gallon (0.86g/ml). Then grams/day is divided by kg of fuel/day to produce fuel specific emissions in grams of pollutant/kg of fuel.

Figure 22 compares the fuel specific NO_x emissions between the Utah 2020 measurements, MOVES3 estimates for Box Elder county Utah and an EMFAC2021 Statewide California emissions estimate. Uncertainties for the Utah measurements are standard error of the mean determined using the daily means. The Utah 1998 chassis model year includes 1998 and older HDV. MOVES3 engine model years have had one year added to them to convert to chassis model year to match the measurements and EMFAC2021 estimates. The measurements have significantly higher NO_x emissions for all model years except for the 2003 and older chassis model years where there is good agreement. The reduction in emissions predicted by the models starting with the 2004 chassis model year vehicles and the subsequent reduction to lower levels after the 2014 chassis model year vehicles do not occur in the Utah measurements. As pointed out previously, within the uncertainties NO_x emissions from the Utah observations are the same for all chassis model years between 2000 and 2013. MOVES3 estimates higher NO_x emissions for 2014 and newer chassis model years when compared with EMFAC2021 due to the inclusion of a significant percentage of Glider (MOVES3 type 49) HDV.

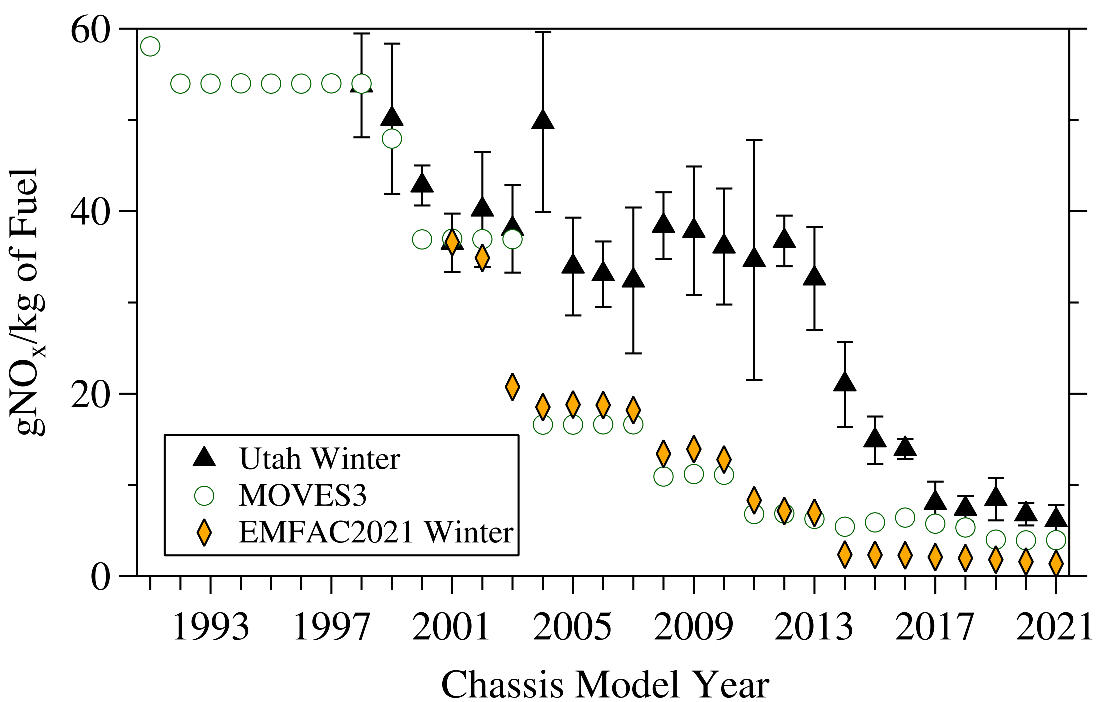

Figure 22. Fuel specific NO_x emissions versus chassis model year for the Utah measurements, the MOVES3 estimates for Box Elder County and EMFAC2021 winter estimates for California. Uncertainties are standard error of the mean calculated using the daily means.

Using the age distribution observed in the Utah measurements we can calculate a mean emissions estimate for each of the model estimates. We use the model estimated emissions factor for each chassis model year and multiply that by the observed chassis model year fraction from the Utah measurements and then sum each product together to produce a mean emissions estimate for each model from a fleet with the same age as the Utah measurements. Table 5 shows the comparisons for the three model estimates. Uncertainties for the Utah measurements are standard error of the mean determined from the daily measurements. Since the EMFAC2021 model does not calculate an emissions factor for any HDV older than chassis model year 2001 we have included all of the vehicles chassis model year 2001 and older vehicles in that last group. We have included the fuel specific NO_x emissions estimates for the EMFAC2021 model for the winter and the summer scenarios. Also EMFAC2021 estimates fuel specific NH_3 emissions and that comparison has been included as well.

Table 5. Comparison of the Utah class 7 & 8 HDV fleet measurements with the model estimated mean emissions.[a]

Data Source	gCO/kg of Fuel	gNO_x/kg of Fuel	gNH_3/kg of Fuel
Utah Measurements	5.8 ± 1.5	18.5 ± 2.0	0.08 ± 0.06
MOVES3	3.3	10.4	N.A.
EMFAC2021 Winter[b]	0.8	7.3	0.33
EMFAC2021 Summer[b]		4.7	0.33

[a]All of the model estimated means have been calculated from their chassis model year emission factors using the Utah measurement fleet model year distribution.
[b]Because EMFAC2021 only models vehicle through model year 2001 this chassis model year includes all of the HDV 2001 and older for the calculated means.

Both of the models can be seen to significantly underestimate the mean NO_x emissions, EMFAC2021 by more than a factor of two and MOVES3 by a factor of 1.8. It is probably reasonable to expect the California model to be lower than the MOVES3 estimates as we would expect a California only fleet to include a higher percentage of low NO_x (less than the current 0.2 gNO_x/bhp-hr) HDV. It should be pointed out that not all of the differences in the means are accounted for by the large differences observed in Figure 22 between the 2004 and 2016 chassis model years because half of the Utah fleet is 2017 and newer and the averaged emission factors for just that chassis model year grouping are 7.4, 4.5 and 1.8 for the Utah, MOVES3 and EMFAC2021 winter respectively.

Figure 23 shows the comparison for fuel specific CO emissions for the two models and the Utah measurements. Uncertainties for the Utah measurements are standard error of the mean determined using the daily means. MOVES3 engine model years have had one year added to them to convert to chassis model year to match the measurements and EMFAC2021 estimates.

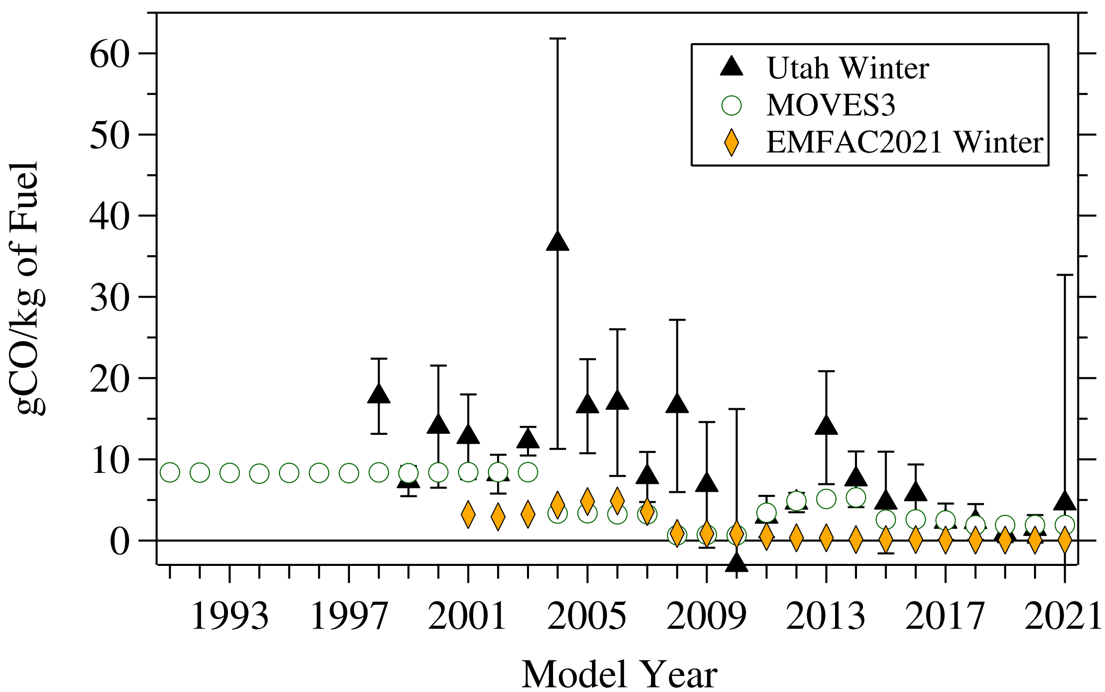

Figure 23. Fuel specific CO emissions versus chassis model year for the Utah measurements, the MOVES3 estimates for Box Elder County and EMFAC2021 winter estimates for California. Uncertainties are standard error of the mean calculated using the daily means.

CO is not usually a pollutant of interest from HDV but we decided to show the comparison as the MOVES3 model does a reasonable job of estimating the observed CO emissions.

Glider HDV in the Utah Fleet. The MOVES3 estimates benefits from the inclusion of a significant number of Glider HDV (increases the mean NO_x emissions by 32%). These are generally new HDV chassis that have had engines install by an after-market supplier. They typically contain older technology engines that may or may not be equipped with particle (DPF) and or NO_x emission after-treatment systems (SCR). MOVES3 estimates that the Glider population in Utah accounts for 3.3% of the fleet and for 2017 & newer HDV they are estimated to account for 4.3% of these vehicles. VIN information for Gliders may not be complete but Peterbilt and Freightliner do mark chassis' as an "Incomplete - Glider" along with the chassis model year. In the Utah measurements we were able to identify 23 Gliders out of 1591 HDV (1.4%) and in the 2017 and newer HDV they account for only 1.1% of the Utah fleet. Both of these values are significantly lower than estimated by the model and if corrected for would further lower the MOVES3 model estimates for NO_x. We can use the age distribution of the Glider's and compare mean NO_x emissions against a similarly aged Utah fleet. The Glider's fuel specific NO_x emissions are 42.3 ± 4.2 and the Utah fleet is 16.5 ± 2.7 gNO_x/kg of fuel or 2.5 times lower than the Glider labeled vehicles. We see a similar difference when we compare the %IR Opacity means as well with the Gliders having a mean %IR Opacity of 2.7 (only 18 valid measurements) versus a mean of 0.6 for the Utah fleet.

We have not removed the Glider's from the Utah fleet in this comparison as they are a very small minority of the 2011 and newer fleet (22/1236). However, if they were to be eliminated we find that they would account for about 4.5% (12.2 vs 11.6 gNOx/kg of fuel) of the 2011 & newer NO_x emissions. Figure 24 shows the 22 2011 & newer Gliders fuel specific NO_x emissions overlaid on the on the fuel specific NO_x emissions for the Utah fleet. The uncertainties are standard error of the mean calculated using the daily means. Fourteen of the 22 vehicles identified as Gliders are registered in Utah. It should also be pointed out that this identification has relied on the VIN information provided by the manufacturers and may not fully capture the number of trucks that may have been retrofit with a different engine later in its service life.

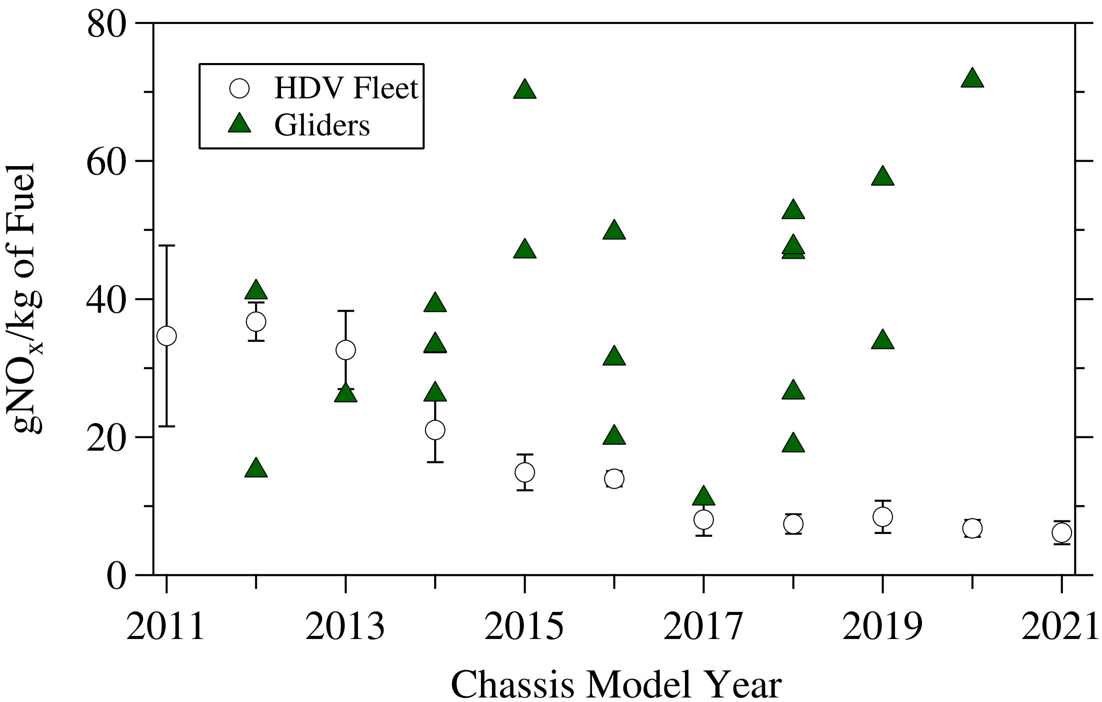

Figure 24. Fuel specific NO_x emissions versus chassis model year for chassis model years 2011 and newer compared to individual NO_x emissions from HDV labeled as Gliders. Uncertainties are standard error of the mean calculated using the daily means.

Cummins Voluntary Recall. As previously mentioned the decreases in NO_x emissions versus chassis model year do not occur as soon as in other warm weather measurements, in particular with the introduction of NO_x after-treatment systems in 2011 to 2013 HDV (see Figure 12). We know that emission deterioration occurred at a higher rate than anticipated in these chassis model year vehicles and part of that unexpected deterioration resulted in a major manufacturer (Cummins) voluntarily recalling a large number of HDV of particular engine families from chassis model years 2011 - 2016 because of SCR problems.[38] We do not have enough information to precisely remove the effected vehicles nor do we know what the repair status of any of the recalled vehicles is. So we have taken a simplistic and broad brushed look at the possible effects of this deterioration on NO_x emissions by comparing the Utah fleet with and without all Cummins engines for the effected model years.

Figure 25 is a graph of fuel specific NO_x emissions versus chassis model year for all of the class 7 & 8 HDV observed during the Utah campaign for 2011 and newer chassis model year vehicles compared against the same fleet but with all of the HDV with Cummins engines removed. Within the uncertainties there are no statistically significant changes, however, chassis model years 2012 and 2013 show noticeable emission reductions. This again suggests that NO_x emissions deterioration is an important component in the increased emissions observed in these early model year vehicles first equipped with NO_x after-treatment systems.

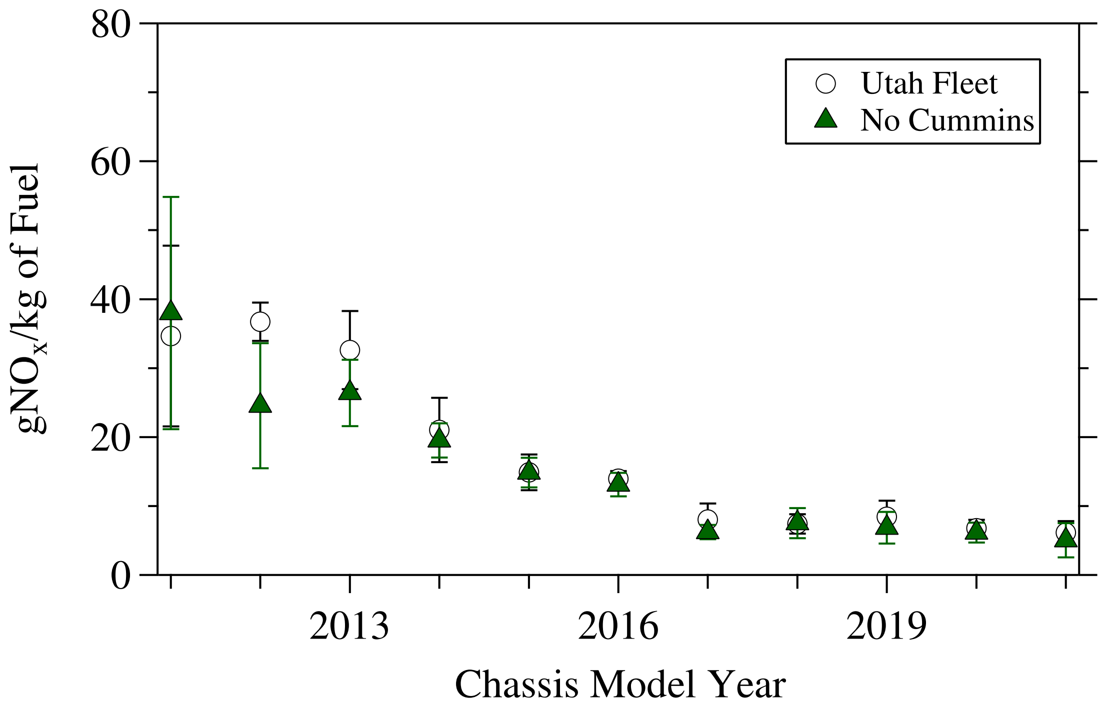

Figure 25. Fuel specific NO_x emissions versus chassis model year for chassis model years 2011 and newer compared against that fleet with all HDV powered by Cummins engines removed. Uncertainties are standard error of the mean calculated using the daily means.

It is not simple to take the model under predictions shown in Table 5 and instantly double the NO_x inventory as the emission factors are only one piece used to calculate the inventory. The activity component, whether it is fuel burned or in the case of MOVES3 VMT/day has to be incorporated on a model year basis for the final inventory calculation. It is safe to say that having emission factors which are a factor of 1.8 higher than the MOVES3 prediction can only increase the Salt Lake City winter NO_x inventory. Since the measured concentration of aerosol nitrate in the Salt Lake City area suggested the possibility of the NO_x emissions inventory under predicting the total emissions, this data supports that contention.

Figure 26 is a bar chart that compares the MOVES3 default percent of Vehicle Miles Traveled (VMT) for types 46, 47 and 49 by chassis model year against the observed age distribution of the Utah measurements. VMT is a reasonable surrogate to compare against the observed age distribution as the probability of our measuring a particular chassis model year increases proportionally to the amount of miles driven. The shape of the two distributions are somewhat similar, however, MOVES3 significantly underestimates the 1 to 3 year old HDV (chassis model years 2018 - 2020, 27% versus 36.4%) and overestimates the 5 to 13 year old HDV (chassis model years 2008 - 2016, 43.8% versus 31%).

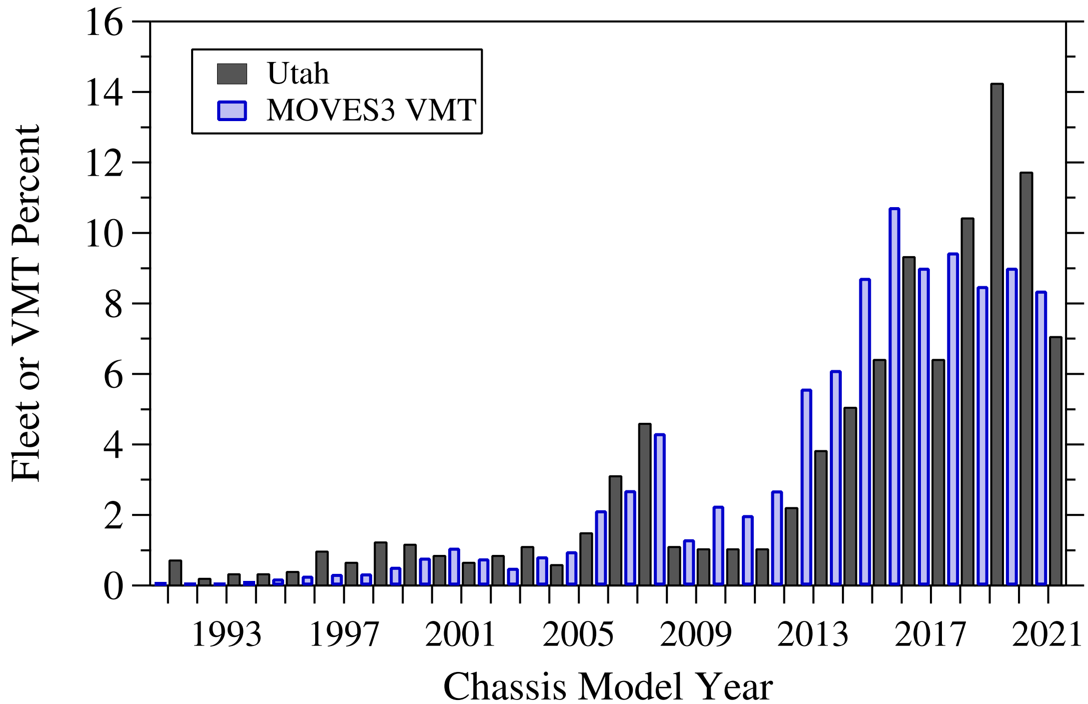

Figure 26. A bar chart comparing the Utah fleet percent or the MOVES3 VMT percent (types 46, 47 and 49) by chassis model year.

Conclusions

Over parts of six days (Sunday Dec. 6 to Friday Dec. 11, 2020) the University of Denver collected exhaust emission measurements with the FEAT remote sensor at the southbound Perry Port of Entry along I-15 just south of Brigham City Utah. The remote sensor was operated in an elevated configuration for ~22 hours to capture exhaust plumes from heavy-duty vehicles with elevated exhaust stacks and at ground level for ~10 hours to sample trucks with ground level pipes. The majority of measurements were collected during daylight hours (~28 hrs.), however, on two evenings some measurements were collected after dark.

This campaign resulted in the successful measurement of 1694 vehicles, the majority of which were class 7 & 8 heavy-duty vehicles (1591) and the remaining 103 measurements from medium-duty vehicles. Vehicles from 37 different states and Canada were sampled with the largest numbers from Utah (35.5%) and Idaho (13.9%). For the 1591 HDV the mean CO, HC, NO, NH_3, NO_2 and NO_x emissions were 5.8 ± 1.5 gCO/kg of fuel, -0.08 ± 0.07 gHC/kg of fuel, 11.5 ± 1.3 gNO/kg of fuel, 0.08 ± 0.06 gNH_3/kg of fuel, 0.67 ± 0.09 gNO_2/kg of fuel, 18.5 ± 2.0 gNO_x/kg of fuel and 0.6 ± 0.1 %IR Opacity respectively. The average chassis model year was 2014.2 and the Utah plated vehicles were 2.4 model years older than the out of state fleet. Outside air temperatures were recorded at the site at 5 minute intervals during all 6 days of sampling with ambient temperatures ranging from -7 to 10°C with an average of 3.8°C for the HDV measurements. Exhaust thermographs were taken with an infrared camera (Thermovision A20, FLIR Systems) of the elevated exhaust pipes of the trucks and pipe temperatures were estimated to be 92 °C.

Fuel specific NO_x emissions were significantly higher than the most recent warm weather measurements collected at a weigh station in California in 2017. While it's difficult to unequivocally ascribe the increased NO_x emissions to a specific cause for a single data set the analysis of the oldest and newest trucks in the fleets suggests a temperature effect that increases NO_x emissions between 8 and 25%. Using the IR thermographs from only the elevated exhaust pipes we found that the estimated mean pipe temperature of 92° in Utah was 18°C colder than similar readings from the California study (110°C) again suggesting colder temperatures for the exhaust after-treatment systems. In addition we did not find that the lower temperatures resulted in an increase in the number of SCR systems that were completely inactive but we believe that the reduced temperatures likely lowers the NO_x conversion efficiencies and increases NO_x emissions in trucks found in the middle percentiles. Increases in the NO_x emissions of the models with the oldest SCR systems (chassis model years 2011 - 2013) appear to most likely be caused by significant emissions deterioration in their after-treatment systems.

Comparison of the HDV NO_x emission measurements with the newest U.S. Environmental Protection Agencies MOVES3 model found significant differences in 2004 and newer chassis model year vehicles. Using the Utah age distribution and the MOVES3 emission factors by chassis model year resulted in mean NO_x emissions of 18.5 ± 2.0 and 10.4 for the Utah measurements and the MOVES3 estimate respectively. The MOVES3 model was run for Utah in December of 2020 but note that MOVES3 does not make a distinction between summer and winter. MOVES3 model mean NO_x emissions is helped by including a larger number of "Glider" vehicles (3.3% of the MOVES3 fleet versus 1.5% observed in the Utah fleet) which have significantly higher NO_x emissions. The factor of 1.8 under prediction by the model likely means that the winter NO_x inventory for the Salt Lake region is also under estimated. The MOVES3 estimates are quite good for 2003 & older chassis model year vehicles but going forward in age the MOVES3 estimates drop significantly faster than observed in the in-use fleet at Perry. A comparison between the Utah measurements and MOVES3 estimates for fuel specific CO emissions results in 5.8 ± 1.5 and 3.3 gCO/kg of fuel for the measurements and the MOVES3 estimates respectively.

We also investigated the EMFAC2021 model that the State of California produces and it does allow for a winter scenario but the California heavy-duty fleet is significantly different than what we would expect to find nationally and it predicted a gNO_x/kg of fuel of 7.3. EMFAC2021 only models back to chassis model year 2001 vehicles whose emissions are predicted quite well but it also follows a similar path as MOVES3 and estimates that NO_x emissions should decrease at a much faster rate than observed. For the newest model year vehicles EMFAC2021 estimates NO_x emissions that are lower than MOVES3 in large part due to the incorporation of Glider vehicles in the MOVES3 fleet. Comparison of fuel specific NO_x emissions estimated by EMFAC2021 in the winter or summer does result in mean NO_x emissions being 34% higher during the winter scenario.

We also were able to show that some of the observed NO_x emission deterioration observed in the 2011 - 2013 chassis model year vehicles is likely related to the voluntary recall of Cummins engines and after-treatment systems for defective SCR systems. In addition there is a very small population of HDV identified as "Gliders" (22/1200) which have significantly higher NO_x and %IR Opacity than similar chassis model year vehicles.

Acknowledgments

The authors would like to acknowledge the Utah Department of Environmental Quality for the funding and Christopher Pennell for the help with numerous queries and project management. The Utah Department of Transportation for allowing us to work at the Perry Port of Entry and in particular Mr. Howard Trexler and Ms. Donnetta Ford for the help in making that happen. Ms. Stacy Hammond and Mr. Brian Himes for the help matching the Utah and Idaho license plates to state registration records. Rick McKeague III for tirelessly running and rerunning the MOVES3 model. Ms. Annette Bishop for help in acquiring repair parts for a computer that did not like the cold during the data collection campaign and Mr. Jim Moini of http://moini.net for sharing his license plate collection and the invaluable resource it is for this type of work.

Literature Cited

1. Markets Insider, 11 incredible facts about the $700 billion U.S. trucking industry. **2019**.

2. Health Effects Institute, *Traffic-related Air Pollution: A Critical Review of the Literature on Emissions, Exposure, and Health Effects. A Special Report of the HEI Panel of the Health Effects of Traffic-related Air Pollution*; 2010; http://pubs.healtheffects.org/getfile.php?u=553.

3. IARC: Diesel engine exhaust carcinogenic. Press Release 213. International Agency for Research on Cancer. http://www.iarc.fr/en/media-centre/pr/2012/pdfs/pr213_E.pdf (Aug. 2016).

4. Office of Environmental Health Hazard Assessment; California Environmental Protection Agency, *Part B: Health risk assessment for diesel exhaust*; Sacramento, 1998; http://www.arb.ca.gov/regact/diesltac/partb.pdf.

5. Adler, J., Ceramic diesel particulate filters. *International Journal of Applied Ceramic Technology* **2005,** 2, (6), 429-439.

6. Koebel, M.; Elsener, M.; Kleemann, M., Urea-SCR: A promising technique to reduce NOx emissions from automotive diesel engines. *Catal. Today* **2000,** 59, (3-4), 335-345.

7. Truck and Bus Regulation. California Environmental Protection Agency Air Resources Board. https://ww2.arb.ca.gov/our-work/programs/truck-and-bus-regulation (May 2021).

8. Regulation to reduce emissions of diesel particulate matter, oxides of nitrogen and other criteria pollutants, from in-use heavy-duty diesel-fueled vehicles. In *California Code of Regulations*; Title 13, Section 2025, 2008.

9. Haugen, M. J.; Bishop, G. A., Long-Term Fuel-Specific NOx and Particle Emission Trends for In-Use Heavy-Duty Vehicles in California. *Environ. Sci. Technol.* **2018,** 52, (10), 6070-6076, DOI: 10.1021/acs.est.8b00621.

10. Bishop, G. A.; Schuchmann, B. G.; Stedman, D. H.; Lawson, D. R., Emission Changes Resulting from the San Pedro Bay, California Ports Truck Retirement Program. *Environ. Sci. Technol.* **2012,** 46, 551-558, DOI: 10.1021/es202392g.

11. Preble, C. V.; Cados, T. E.; Harley, R. A.; Kirchstetter, T. W., In-Use Performance and Durability of Particle Filters on Heavy-Duty Diesel Trucks. *Environ. Sci. Technol.* **2018,** 52, (20), 11913-11921, DOI: 10.1021/acs.est.8b02977.

12. Preble, C. V.; Dallmann, T. R.; Kreisberg, N. M.; Hering, S. V.; Harley, R. A.; Kirchstetter, T. W., Effects of Particle Filters and Selective Catalytic Reduction on Heavy-Duty Diesel Drayage Truck Emissions at the Port of Oakland. *Environ. Sci. Technol.* **2015,** 49, (14), 8864-8871, DOI: 10.1021/acs.est.5b01117.

13. Yu, K. A.; McDonald, B. C.; Harley, R. A., Evaluation of Nitrogen Oxide Emission Inventories and Trends for On-Road Gasoline and Diesel Vehicles. *Environ. Sci. Technol.* **2021,** 55, (10), 6655-6664, DOI: 10.1021/acs.est.1c00586.

14. Brown, S. S.; Baasandorj, M.; Franchin, A.; Middlebrook, A. M.; Goldberger, L.; Thornton, J. A.; Dube, W. P.; McDuffie, E. E.; Womack, C.; Fibiger, D. L.; Moravek, A.; Clark, J. C.; Murphy, J. G.; Mitchell, R., Aircraft measurements to characterize polluted winter boundary

layers: Overview of twin otter flights during the Utah Winter Fine Particulate Matter Study. In *AGU Fall Meeting Abstracts*, 2017.

15. Chan, E. A. W.; Gantt, B.; McDow, S., The reduction of summer sulfate and switch from summertime to wintertime PM2.5 concentration maxima in the United States. *Atmos. Environ.* **2018,** 175, 25-32, DOI: https://doi.org/10.1016/j.atmosenv.2017.11.055.

16. Grange, S. K.; Farren, N. J.; Vaughan, A. R.; Rose, R. A.; Carslaw, D. C., Strong Temperature Dependence for Light-Duty Diesel Vehicle NOx Emissions. *Environ. Sci. Technol.* **2019,** 53, (11), 6587-6596, DOI: 10.1021/acs.est.9b01024.

17. McDonald, B. C.; McKeen, S. A.; Cui, Y. Y.; Ahmadov, R.; Kim, S.-W.; Frost, G. J.; Pollack, I. B.; Peischl, J.; Ryerson, T. B.; Holloway, J. S.; Graus, M.; Warneke, C.; Gilman, J. B.; de Gouw, J. A.; Kaiser, J.; Keutsch, F. N.; Hanisco, T. F.; Wolfe, G. M.; Trainer, M., Modeling Ozone in the Eastern U.S. using a Fuel-Based Mobile Source Emissions Inventory. *Environ. Sci. Technol.* **2018,** 52, (13), 7360-7370, DOI: 10.1021/acs.est.8b00778.

18. Hall, D. L.; Anderson, D. C.; Martin, C. R.; Ren, X.; Salawitch, R. J.; He, H.; Canty, T. P.; Hains, J. C.; Dickerson, R. R., Using near-road observations of CO, NOy, and CO2 to investigate emissions from vehicles: Evidence for an impact of ambient temperature and specific humidity. *Atmos. Environ.* **2020,** 232, 117558, DOI: https://doi.org/10.1016/j.atmosenv.2020.117558.

19. Li, X.; Dallmann, T. R.; May, A. A.; Presto, A. A., Seasonal and long-term trend of on-road gasoline and diesel vehicle emission factors mesured in traffic tunnels. *Applied Science* **2020,** 10, 2458, DOI: 10.3390/app10072458.

20. Saha, P. K.; Khlystov, A.; Snyder, M. G.; Grieshop, A. P., Characterization of air pollutant concentrations, fleet emission factors, and dispersion near a North Carolina interstate freeway across two seasons. *Atmos. Environ.* **2018,** 177, 143-153, DOI: https://doi.org/10.1016/j.atmosenv.2018.01.019.

21. Bishop, G. A.; Starkey, J. R.; Ihlenfeldt, A.; Williams, W. J.; Stedman, D. H., IR long-path photometry, A remote sensing tool for automobile emissions. *Anal. Chem.* **1989,** 61, 671A-677A, DOI: 10.1021/ac00185a746.

22. Burgard, D. A.; Dalton, T. R.; Bishop, G. A.; Starkey, J. R.; Stedman, D. H., Nitrogen dioxide, sulfur dioxide, and ammonia detector for remote sensing of vehicle emissions. *Rev. Sci. Instrum.* **2006,** 77, (014101), 1-4, DOI: 10.1063/1.2162432.

23. Guenther, P. L.; Stedman, D. H.; Bishop, G. A.; Bean, J. H.; Quine, R. W., A hydrocarbon detector for the remote sensing of vehicle emissions. *Rev. Sci. Instrum.* **1995,** 66, 3024-3029.

24. Popp, P. J.; Bishop, G. A.; Stedman, D. H., Development of a high-speed ultraviolet spectrometer for remote sensing of mobile source nitric oxide emissions. *J. Air Waste Manage. Assoc.* **1999,** 49, 1463-1468, DOI: 10.1080/10473289.1999.10463978.

25. Burgard, D. A.; Bishop, G. A.; Stadtmuller, R. S.; Dalton, T. R.; Stedman, D. H., Spectroscopy applied to on-road mobile source emissions. *Appl. Spectrosc.* **2006,** 60, 135A-148A, DOI: 10.1366/000370206777412185.

26. Singer, B. C.; Harley, R. A.; Littlejohn, D.; Ho, J.; Vo, T., Scaling of infrared remote sensor hydrocarbon measurements for motor vehicle emission inventory calculations. *Environ. Sci. Technol.* **1998,** 32, 3241-3248, DOI: 10.1021/es980392y.

27. Broering, L. C. Diesel engine emissions reduction history and future prospects, In *National Conference of State Legislatures*, 1997;

28. Haugen, M. J.; Bishop, G. A., Repeat Fuel Specific Emission Measurements on Two California Heavy-Duty Truck Fleets. *Environ. Sci. Technol.* **2017,** 51, (7), 4100-4107, DOI: 10.1021/acs.est.6b06172.

29. Emissivity Coefficients of some common Materials. The Engineering ToolBox. http://www.engineeringtoolbox.com/emissivity-coefficients-d_447.html (January 2013).

30. Bishop, G. A.; Schuchmann, B. G.; Stedman, D. H., Heavy-Duty Truck Emissions in the South Coast Air Basin of California. *Environ. Sci. Technol.* **2013,** 47, (16), 9523-9529, DOI: 10.1021/es401487b.

31. Haugen, M. J.; Bishop, G. A.; Thiruvengadam, A.; Carder, D. K., Evaluation of Heavy- and Medium-Duty On-Road Vehicle Emissions in California's South Coast Air Basin. *Environ. Sci. Technol.* **2018,** 52, (22), 13298-13305, DOI: 10.1021/acs.est.8b03994.

32. Bishop, G. A.; Morris, J. A.; Stedman, D. H.; Cohen, L. H.; Countess, R. J.; Countess, S. J.; Maly, P.; Scherer, S., The effects of altitude on heavy-duty diesel truck on-road emissions. *Environ. Sci. Technol.* **2001,** 35, 1574-1578, DOI: 10.1021/Es001533a.

33. Stanton, D., Systematic Development of Highly Efficient and Clean Engines to Meet Future Commercial Vehicle Greenhouse Gas Regulations. *SAE Int. J. Engines* **2013,** 6, 1395-1480, DOI: 10.4271/2013-01-2421.

34. U. S. Environmental Protection Agency, *Exhaust emission rates for heavy-duty on-road vehicles in MOVES3*; 2020; https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1010MC2.pdf.

35. Modeling and Inventories; MOVES (Motor Vehicle Emission Simulator). U. S. Environmental Protection Agency. http://www.epa.gov/otaq/models/moves/ (October, 2014).

36. FEAT Math II. Bishop, G. A. http://www.feat.biochem.du.edu/assets/reports/FEAT_Math_II.pdf (December 2017).

37. EMFAC Emissions Database. California Environmental Protection Agency; Air Resources Board. http://www.arb.ca.gov/emfac/ (July, 2019).

38. Cummins to recall 500,000 medium- and heavy-duty trucks due to SCR issue. Fleet Equipment. https://www.fleetequipmentmag.com/cummins-to-recall-500000-medium-and-heavy-duty-trucks-due-to-scr-issue/.

APPENDIX A: FEAT criteria to render a reading "invalid".

Invalid :

1) insufficient plume to rear of vehicle relative to cleanest air observed in front or in the rear; at least five, 10ms >160ppm CO_2 or >400 ppm CO. (0.2 %CO_2 or 0.5% CO in an 8 cm cell. This is equivalent to the units used for CO_2 max.). For HDDV's this often occurs when the vehicle shifts gears at the sampling beam.

2) excessive error on CO/CO_2 slope, equivalent to $\pm$20% for CO/CO_2 > 0.069, 0.0134 CO/CO_2 for CO/CO_2 < 0.069.

3) reported CO/CO_2 , < -0.063 or > 5. All gases invalid in these cases.

4) excessive error on HC/CO_2 slope, equivalent to $\pm$20% for HC/CO_2 > 0.0166 propane, 0.0033 propane for HC/CO_2 < 0.0166.

5) reported HC/CO_2 < -0.0066 propane or > 0.266. HC/CO_2 is invalid.

6) excessive error on NO/CO_2 slope, equivalent to $\pm$20% for NO/CO_2 > 0.001, 0.002 for NO/CO_2 < 0.001.

7) reported NO/CO_2 < -0.00465 or > 0.0465. NO/CO_2 is invalid.

8) excessive error on SO_2/CO_2 slope, ± 0.0134 SO_2/CO_2.

9) reported SO_2/CO_2 , < -0.00053 or > 0.0465. SO_2/CO_2 is invalid.

10) excessive error on NH_3/CO_2 slope, ± 0.00033 NH_3/CO_2.

11) reported NH_3/CO_2 < -0.00053 or > 0.0465. NH_3/CO_2 is invalid.

12) excessive error on NO_2/CO_2 slope, equivalent to $\pm$20% for NO_2/CO_2 > 0.00133, 0.000265 for NO_2/CO_2 < 0.00133.

13) reported NO_2/CO_2 < -0.0033 or > 0.0465. NO_2/CO_2 is invalid.

Speed/Acceleration valid only if at least two blocks and two unblocks in the time buffer and all blocks occur before all unblocks on each sensor and the number of blocks and unblocks is equal on each sensor and 100mph>speed>5mph and 14mph/s>accel>-13mph/s and there are no restarts, or there is one restart and exactly two blocks and unblocks in the time buffer.

APPENDIX B: Explanation of the Utah_20.dbf database.

The Utah_20.dbf is a Microsoft FoxPro database file, and can be opened by any version of MS FoxPro. These files can be read by a number of other database management and spreadsheet programs as well, and is available from www.feat.biochem.du.edu. The grams of pollutant/kilogram of fuel consumed are calculated assuming that diesel fuel has 860 grams of carbon per kilogram of fuel and natural gas has 750 grams of carbon per kilogram of fuel. The following is an explanation of the data fields found in this database:

License	Anonymized license plate which preserves duplicate measurements.
State	State or country (CN=Canada) license plate issued by.
Date	Date of measurement, in standard format.
Time	Time of measurement, in standard format.
Co_co2	Measured carbon monoxide / carbon dioxide ratio
Co_err	Standard error of the CO/CO_2 measurement.
Hc_co2	Measured hydrocarbon / carbon dioxide ratio (propane equivalents).
Hc_err	Standard error of the HC/CO_2 measurement.
No_no2	Measured nitric oxide / carbon dioxide ratio.
No_err	Standard error of the NO/CO_2 measurement.
So2_co2	Measured sulfur dioxide / carbon dioxide ratio.
So2_err	Standard error of the SO_2/CO_2 measurement.
Nh3_co2	Measured ammonia / carbon dioxide ratio.
Nh3_err	Standard error of the NH_3/CO_2 measurement.
No2_co2	Measured nitrogen dioxide / carbon dioxide ratio.
No2_err	Standard error of the NO_2/CO_2 measurement.
Opacity	IR Opacity measurement, in percent.
Opac_err	Standard error of the opacity measurement.
Restart	Number of times data collection is interrupted and restarted by a close-following vehicle, or the rear wheels of tractor trailer.
Hc_flag	Indicates a valid hydrocarbon measurement by a “V”, invalid by an “X”.
No_flag	Indicates a valid nitric oxide measurement by a “V”, invalid by an “X”.
So2_flag	Indicates a valid sulfur dioxide measurement by a “V”, Invalid by an “X”.
Nh3_flag	Indicates a valid ammonia measurement by a “V”, Invalid by an “X”.
No2_flag	Indicates a valid Nitrogen dioxide measurement by a “V”, Invalid by an “X”.
Opac_flag	Indicates a valid opacity measurement by a “V”, invalid by an “X”.

CO2_max	Reports the highest absolute concentration of carbon dioxide measured by the remote sensor over an 8 cm path; indicates plume strength.
Speed_flag	Indicates a valid speed measurement by a "V", an invalid by an "X", and slow speed (excluded from the data analysis) by an "S".
Speed	Measured speed of the vehicle, in mph.
Accel	Measured acceleration of the vehicle, in mph/s.
Tag_name	File name for the digital picture of the vehicle.
Exh_temp	Temperature in degree C of elevated exhaust pipes from IR thermograph.
Make	Manufacturer of the vehicle.
Year	Model year of the vehicles chassis.
Vin	Vehicle identification number.
Model	Vehicle model from registration information
Zip	Zip code of location where vehicle resides from registration information.
V_cabtype	VIN decoded tractor cab type.
V_bodycl	VIN decoded tractor body class.
V_cyl	VIN decoded number of engine cylinders.
V_displ	VIN decoded engine displacement in liters.
V_engmod	VIN decoded engine model.
V_fuel	VIN decoded fuel type.
V_gvwr	VIN decoded gross vehicle weight class.
V_wtclass	VIN decoded weight class number.
V_manufact	VIN decoded vehicle manufacturer.
V_model	VIN decoded model information.
V_series	VIN decoded vehicle series information.
V_type	VIN decoded vehicle type.
V_enghp	VIN decoded engine horsepower.
V_engman	VIN decoded engine manufacturer.
CO_gkg	Grams of CO per kilogram of fuel consumed.
HC_gkg	Grams of HC per kilogram of fuel consumed.
NO_gkg	Grams of NO per kilogram of fuel consumed.
NH3_gkg	Grams of NH_3 per kilogram of fuel consumed.
NO2_gkg	Grams of NO_2 per kilogram of fuel consumed.

NOx_gkg	Grams of NO_x per kilogram of fuel consumed.
FEAT	Location of FEAT (High or Low)
Temp_5c	Site temperature 5 minute intervals (deg C).
Temp_15c	Temperature averaged over the preceding 3 5 minute readings (deg C).
STP	Scaled tractive power in skw/tonne.

APPENDIX C: MOVES3 Information

MOVES3 Vehicle, Fuel and Other Parameters Used

regClassID	regClassDesc
46	Class 6 and 7 Trucks (19,500 lbs < GVWR > 33,000 lbs)
47	Class 8a and 8b Trucks (GVWR>33,000 lbs)
49	Glider Vehicles (see EPA-420-F-15-904)

regClassID	sourceTypeID	sourceTypeName
46, 47	61	Combination Short-haul Truck
46, 47	62	Combination Long-haul Truck

fueltypeID	fuelTypeDesc	humidityCorrectionCoeff	fuelDensity
2	Diesel Fuel	0.0026	3167

Model was run for December 5, 2020 and for Box Elder County (countyid=49003) and output was generated for hot running emissions for an urban restricted access road type (type 4).

Notes: MOVES3 reports NO as grams of NO_2 and vehicle model year output is for engine model year. We have added one year to the engine model years to convert to chassis model year.

To convert gram to moles we have used the molecular weights of 44 grams/mole for CO_2, 28 grams/mole for CO, 46 grams/mole for both NO and NO_x and because MOVES3 reports HC emissions as measured by a flame ionization detector we have used 12 grams/mole to convert the Total Gas HC to moles of HC.

The molar ratios to CO_2 have been converted to fuel specific emissions using the same equations that we use to convert the FEAT measured molar ratios. We have used 860 grams Carbon per kilogram of fuel.

$$gCO/kg\ of\ Fuel = (28 * 860 * CO/CO_2) / ((1 + CO/CO_2 + HC/CO_2)*12)$$

$$gNO_x/kg\ of\ Fuel = (46 * 860 * NO_x/CO_2) / ((1 + CO/CO_2 + HC/CO_2)*12)$$

MOVES3 Emission Output Tables.

Emissions in grams/day						
Type 46 and 47 all sources (Class 6, 7, 8a and 8b)						
Chassis MY	Atmospheric CO2	CO	NO as NO2	NOx	Total HC	vmt
1991	216740.05	552.39	3750.28	4011.00	80.79	108.54
1992	105153.70	277.00	1692.12	1809.76	38.61	52.76
1993	148359.31	389.73	2387.65	2553.63	54.54	74.37
1994	265261.53	686.57	4270.66	4567.56	98.18	132.99
1995	402606.20	1055.81	6479.64	6930.10	148.11	201.80
1996	572748.20	1501.86	9218.07	9858.91	210.86	287.43
1997	666825.50	1740.00	10733.09	11479.24	245.64	333.86
1998	700447.54	1847.20	11270.07	12053.54	256.44	350.45
1999	1074156.20	2809.71	15355.29	16422.74	394.62	536.85
2000	1600960.40	4237.68	17618.53	18843.39	584.48	799.68
2001	2168291.80	5759.80	23899.53	25560.94	790.67	1083.21
2002	1556745.44	4128.80	17143.31	18335.13	568.46	778.24
2003	1020624.26	2706.52	11238.36	12019.63	372.74	510.21
2004	1679552.00	1579.14	8293.16	8869.69	308.16	838.79
2005	1966986.08	1846.31	9710.31	10385.36	360.39	981.59
2006	4317069.60	4065.92	21321.10	22803.32	793.17	2157.03
2007	5463122.00	5149.61	26984.33	28860.24	1004.47	2731.09
2008	8709783.94	1683.65	23056.82	30179.03	328.54	4358.90
2009	2607316.40	508.39	6914.34	9050.18	99.09	1315.71
2010	4503642.00	877.18	11943.10	15632.38	170.98	2270.63
2011	3926187.00	4344.27	4651.15	7830.22	273.60	1985.83
2012	5107612.00	8149.89	5374.35	9047.73	206.35	2612.80
2013	10698583.00	17778.79	10860.09	18283.00	395.10	5478.38
2014	11606514.00	20143.02	9349.15	15739.26	292.13	5952.42
2015	14632346.00	12160.33	12087.24	20348.92	401.62	8408.63
2016	17734481.00	14924.17	14750.89	24833.16	484.54	10185.56
2017	15125665.00	12218.61	12174.52	20495.75	404.98	8702.10
2018	14874038.00	9122.06	7905.71	13309.31	332.43	8883.38
2019	13449186.00	8367.86	7107.59	11965.64	293.24	8292.47
2020	14300697.00	8898.39	7558.46	12724.69	311.84	8817.56
2021	13271720.00	8258.85	7015.49	11810.59	289.45	8183.14

Notes: All emission species are in grams/day and NO is not grams of NO but grams of NO_2. Atmospheric CO_2 is tailpipe CO_2. MOVES3 outputs engine model year. We have added 1 year to each engine model year to approximate chassis model year in the report.

Emissions in grams/day						
Type 49 (Gliders)						
Chassis MY	Atmospheric CO2	CO	NO as NO2	NOx	Total HC	vmt
2009	23761.63	65.57	214.72	281.05	8.85	11.98
2010	28792.57	79.43	260.17	340.53	10.72	14.51
2011	66303.59	182.86	465.77	784.13	24.67	33.38
2012	215506.60	594.45	1513.96	2548.76	80.20	108.54
2013	313811.80	865.74	2204.64	3711.51	116.80	158.11
2014	424973.60	1172.68	2985.75	5026.51	158.22	214.23
2015	703837.20	2004.80	4981.11	8385.70	273.98	387.56
2016	1150590.00	3277.34	8142.83	13708.44	447.88	633.57
2017	710318.00	2023.37	5027.06	8463.05	276.53	391.22
2018	1163405.00	1449.90	8233.68	13861.44	198.16	640.79
2019	476374.90	599.15	3377.62	5686.23	82.56	270.96
2020	476311.80	599.07	3377.19	5685.50	82.54	270.93
2021	446479.60	561.54	3165.66	5329.40	77.37	253.95